Tree hollows in Tasmanian *Eucalyptus obliqua* forest and their use by vertebrate fauna

By

Amelia Jane Koch

B.Sc. (U.Melb.) (Hons.) (Griffith)

October 2007

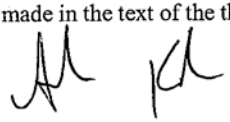
A thesis submitted in total fulfilment of the requirements for the degree of Doctor of Philosophy at the University of Tasmania

School of Geography and Environmental Studies

Hobart, Tasmania

Statement of Originality

This thesis contains no material which has been accepted for the award of any other degree or diploma in any tertiary institution, and to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference is made in the text of the thesis.



Amelia Koch

6/11/07

Date

Statement on authority of access

This thesis may be made available for loan and limited copying in accordance with the *Copyright Act 1968*.

Acknowledgments

First, I would like to give a huge thank you to my three supervisors, Sarah Munks, Don Driscoll and Jamie Kirkpatrick for all their help and support throughout my candidature. I feel extremely lucky to have had such excellent supervision. I really appreciated their concern for both my academic and mental wellbeing. To all of you, I am extremely grateful. The field and lab work could not have been done without the help of my trusty assistant Chris Spencer who was willing to venture out on last-minute notice under any conditions, do long days and still be willing to camp in the rain or drive me home. Other intermittent field assistants to whom I am also grateful are David Wilson and David Allen. Thank you. I would also like to express a deep gratitude to all the forestry workers, at all levels, who assisted me with my project. This work could not have been done without their cooperation. In particular, thanks to the contractors who were happy for a female to venture on to their coupe and went out of their way to help me.

I would like to acknowledge the assistance of advice and information from Eric Woehler and Mark Holdsworth. I would also like to acknowledge the bird distribution data supplied to me by Birds Australia and some data on tree rings supplied by Julie von Platen. Thanks also to Julie and Kathy Allen for counting some tree rings for me. Thanks to David Wilson for producing many versions of the maps of hollow-using species distribution in Tasmania. The staff and students of the Department of Geography and Environmental Studies collectively provided important support, but special thanks to Darren Turner and Denis Charlesworth. Further thanks are to all those who helped me with my analysis. On top of my supervisors, they are Ross Corkrey, Simon Wotherspoon, Michael McCarthy and Chris Wilcox. To all of you I am extremely grateful.

A lot of people contributed to Chapter 2 and their help was very much appreciated. Firstly, a large thanks to Mark Holdsworth for checking the species distribution maps and for his comments on the entire manuscript and all the species therein. Thanks to Stewart Blackhall for his expertise on ducks in terms of checking the distribution maps and editing the text on ducks. Thanks to Chris Spencer for checking the distribution maps of all species. Thanks to Raymond Brereton for discussions on Tasmanian bats and for checking the text on bats in this paper. Thanks to Nick Mooney for reading the sections on Tasmanian owls. Thanks to Matt Webb for comments on the swift parrot, masked owl and the yellow-tailed black cockatoo.

This research was made possible because of an Australian Postgraduate Award. The costs of field and lab work were met by funding from the W.V. Scott Fund, the Tasmanian Forest Practices Authority, Forestry Tasmania, Holsworth Wildlife Research Grant and The University of Tasmania.

Finally, sincere thanks to my family and friends who have supported and encouraged me and made sure I still have a life and personality outside of the office.

Preface

Primary data used in this thesis were collected by myself and Chris Spencer (and a couple of other volunteers) between 1993 and 1995. Other sources of data were bird distribution data collected by an array of people and supplied by Birds Australia (Chapter 2). Some tree ring data was provided by Julie von Platen and it is acknowledged where this was used (Chapter 3).

I did all the statistical analyses throughout the thesis using the programs specified. Soil and leaf chemistry analysis was done by Denis Charlesworth. Identification of animal hairs was done by Barbara Triggs.

This thesis is written as a series of independent scientific papers. As such there is some repetition between the chapters, particularly in the introduction and methods section. Chapter 4 has been accepted for publication in Forest Ecology and Management. I have submitted for publication a version of chapters 3, 5 and Appendix 1 and intend to submit chapters 2 and 6 for publication with members of my supervisory panel as co-authors. I collected the data, undertook the analyses and wrote these chapters, but each member of my supervisory panel (or other co-authors) made contributions to project design.

Abstract

Tree hollows provide an essential resource for a range of fauna, both in Australia and worldwide. There is concern over the future availability of the hollow resource in forests managed for timber harvesting because the time required for hollows to form is generally longer than the interval between harvest rotations. One of the strategies used to maintain the hollow resource is to retain specific trees to provide habitat for fauna.

The overall aim of this thesis was to assess the tree hollow resource and its use by fauna in Tasmania, in order to inform the revision of management prescriptions for the conservation of the hollow resource in production forest areas. It was also to assess whether different prescriptions are required for different forest types. As part of this work, two methods of studying tree hollows were assessed for their accuracy and utility. To achieve these aims, 391 trees were examined before and after felling in forestry coupes throughout Tasmania. Data were collected on tree growth rings, hollow occurrence and whether the trees were being used by fauna. Estimates of hollow abundance on felled trees were corrected by the amount of tree that had been successfully surveyed. The trees were located at 39 sites in two broad forest types important to the Tasmanian forest industry, wet and dry *Eucalyptus obliqua* forest.

The main factors associated with both hollow presence and abundance were the number of hollows observed on the standing tree, tree diameter and the amount of dead wood in the canopy. While tree age was strongly associated with hollow presence, it had less bearing on hollow abundance. An increase in the maximum size of hollow found in a tree was related to greater tree senescence, as indicated by higher values of the afore-mentioned factors. Although significantly more hollows of all sizes were found in wet forest than drier forest, the age at which trees began to produce hollows was similar between the different forest types. Trees grow slightly more slowly in dry forest than wet, meaning that trees need to be slightly larger in wet forest before they are hollow-bearing. The difference in hollow abundance between the different forest types was largely a result of the history of logging in dry *E. obliqua* forest which has resulted in trees being smaller and younger in the areas studied.

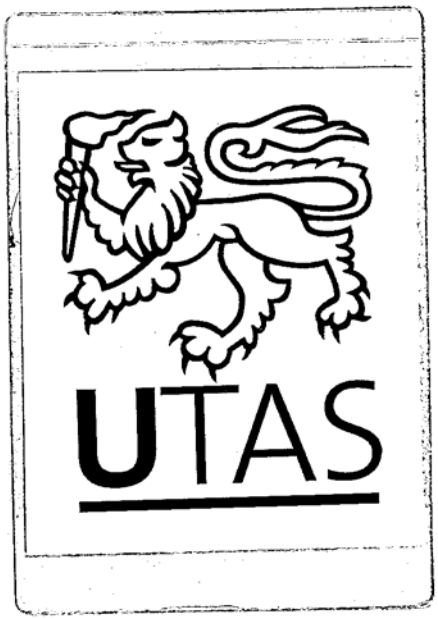
Although hollow-using fauna are a major component of vertebrate fauna in Tasmania, the number of hollow-using species found in Tasmania (42) is at the lower end of the range found in other areas of Australia. The rate at which trees were used by vertebrate fauna in Tasmania was also at the lower end of the scale found in other areas of Australia, with only 28% of hollow-bearing trees examined showing evidence of use. The variables most strongly

related to the use of a tree were hollow abundance, tree size and senescence. The likelihood of a hollow being used increased with hollow size; hollow depth in particular. There was a slight increase in the proportion of trees that showed evidence of use with increasing forest wetness, which corresponded with an increase in the availability of hollows. The rate of hollow use was very low (5%) and it is proposed that this was due to the often shallow and open nature of the hollows observed in this study. This observation may relate to the low numbers of arboreal termites in Tasmania.

It was recommended that trees identified for retention of habitat in logging coupes be selected primarily based on the presence of large hollows. Despite the difficulties in observing hollows in standing trees, ground-based surveys are one of the most effective ways of establishing hollow presence. In wet forest areas, large hollows are most likely to occur in trees that are at least 125 cm in diameter and have at least six visible hollows. In dry forest areas, large hollows are most likely to occur in trees at least 100 cm in diameter with at least six visible hollows.

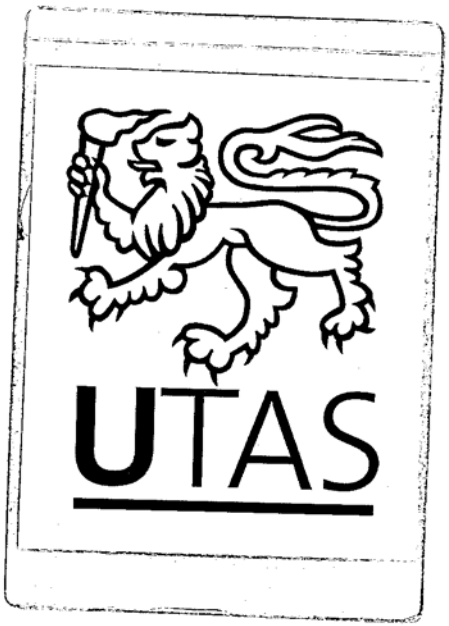
It was estimated that between 8 and 15 trees per hectare were used, on average, by hollow-using fauna. This is significantly more than the rate at which trees are currently being retained in production forests in Tasmania (0.4 to 0.6 per hectare). It is therefore recommended that the rate of retention and type of trees being retained be reviewed in order to more adequately meet the requirements of hollow-using fauna.

Table of contents



List of Figures

List of Tables



Chapter 1
Introduction

The importance of tree hollows

A tree hollow is a semi-enclosed cavity that occurs in trees (Gibbons and Lindenmayer, 2002, p. 1). Tree hollows provide habitat, largely nesting and roosting sites, for many species of birds, mammals, reptiles, amphibians and invertebrates, in many areas of the world (Mackowski, 1984; Lindenmayer *et al.*, 1996a; Webb and Shine, 1997; Martin *et al.*, 2004; Ruczynski and Bogdanowicz, 2005; Walker *et al.*, 2005). Animals are thought to use hollows because they provide protection from predators, can reduce the energy requirements needed to construct a nest, have thermoregulatory advantages and can provide a food resource by sheltering prey (Gibbons and Lindenmayer, 2002). Many species are limited to nesting in tree hollows (e.g. Brown and Wilson, 1984; Smith *et al.*, 1985) but in some parts of the world suitable tree hollows can be in short supply (Berg *et al.*, 1994; Manning *et al.*, 2004; Lohmus and Remm, 2005; Monterrubio-Rico and Escalante-Pliego, 2006). The management of tree hollows can have a direct and important influence on the persistence of some animal species.

Some animal species, such as woodpeckers, can directly excavate hollows (Jackson and Jackson, 2004; Losin *et al.*, 2006). However, in Australia, there are no hollow excavators and so slower processes are involved in hollow formation. In order for hollows to form in eucalypts, there generally needs to be some sort of physical injury, the site of which is enlarged into a cavity by a combination of fire, fungi and termites (Gibbons and Lindenmayer, 2002). In order to provide a useful hollow for fauna, the damaged tree needs to be structurally sound enough that it will remain standing after the onset of hollow formation. Processes such as production forestry, firewood collection, rural dieback, grazing and clearing are all impacting on the availability of tree hollows in Australia (Gibbons and Lindenmayer, 2002, p. 1; Manning *et al.*, 2004), meaning that careful management of this resource is required. Production forestry is of major concern because of the large areas of land impacted and because harvesting rotation times are generally much shorter than the time required to produce a hollow (Lindenmayer *et al.*, 1990c; Ball *et al.*, 1999; Gibbons, 1999). Strategies currently implemented in production forest areas to help maintain a hollow resource generally involve the retention of trees at a range of spatial scales; in large reserves, strips, clumps and as single trees (Keisker, 2000; Wayne *et al.*, 2006).

Retained trees have been shown to be used to aid dispersal, for foraging and breeding in both regenerating forest (Smith, 1985; Wapstra and Taylor, 1998; Mazurek and Zielinski, 2004) and agricultural landscapes (Fischer and Lindenmayer, 2002; Manning *et al.*, 2004). Lindenmayer (1994) reviewed the use of retained strips in Australian temperate wood production forests and found that their use was related to the following factors: the biology, ecology and life history of species; the attributes (width and length) and habitat suitability of the retained strips; the location of sites in the landscape; the type of logging operations and their intensity and pattern at a landscape level; the suitability of habitats for fauna in the areas surrounding retained strips; and the value for fauna of reserves connected by corridors.

Hollow-bearing trees retained within logging coupes for the specific purpose of providing faunal habitat are known as habitat trees. Habitat trees need to be carefully selected to ensure that they meet the requirements of fauna which includes the presence of suitable hollows. To facilitate this selection process, models can be constructed which predict hollow occurrence from site and tree variables. The variables most commonly related to both the occurrence of hollows and use of trees by fauna are measures of tree age, size and senescence. Gibbons and Lindenmayer (2002, p. 75) provide a generic model for selecting a suitable habitat tree, while acknowledging that marked differences between tree species can occur. Firstly, they stated that the tree should contain a number of visible hollows. Secondly, the tree should have signs of damage or physiological stress such as dead branches in the crown. Thirdly, the tree should ideally have a large diameter.

The generic model for selecting a habitat tree proposed by Gibbons and Lindenmayer (2002) is generally supported by a number of studies (Lindenmayer *et al.*, 1990c; Lindenmayer *et al.*, 1996a; Gibbons *et al.*, 2000b; Lindenmayer *et al.*, 2000; Fox *et al.*, 2001; Whitford and Williams, 2002; Manning *et al.*, 2004). However, hollows are often not detected because they are obscured by foliage, or are facing upwards (Whitford, 2002). In wet forest, not only are trees taller, but the understorey is often more dense than is found in dry forest. It is therefore expected that the ability to detect hollows from the ground will vary between forest types.

Wet sclerophyll forests describe those areas dominated by tall eucalypts with a dense understorey of broad-leaved shrubs which prevent continuous regeneration of shade intolerant species such as eucalypts (Wells and Hickey, 1999). Wet forests generally occur in areas of high rainfall and infrequent fire, with regeneration generally occurring after these stand-replacing fires (Wells and Hickey, 1999). Dry sclerophyll forests are dominated by eucalypts and, as a result of frequent disturbance by fire, typically have a multi-layered shrub understorey dominated by xeromorphic species (Duncan, 1999). There is no sharp transition

between wet and dry sclerophyll forests, but a gradual transition as water availability increases (Jackson, 1999).

Eucalypts often grow faster in the more productive wet forests (Bowman and Kirkpatrick, 1984) and can potentially attain a greater size and age (see Banks, 1997) than trees in dry forest. Both tree size and age are related to hollow occurrence and tree use. The distribution of faunal species can be related to factors such as the understorey species present (Lindenmayer *et al.*, 1991) and the nutritional value of the soil and leaves at the site (Braithwaite *et al.*, 1983), factors which again vary between forest types. Furthermore, the rate of disturbance by fire, which promotes hollow formation (Inions *et al.*, 1989), differs between forest types (Bowman and Kirkpatrick, 1984; Gibbons and Lindenmayer, 2002). As a consequence of the difference in fire frequency and intensity, Gibbons and Lindenmayer (2002, p. 119) have postulated that "the number of hollows in wet sclerophyll forests will vary more widely, but on a longer cycle than dry sclerophyll forests". They have also proposed that the rate of hollow loss is likely to be greater in wet forest than in dry forest due to higher rates of decay (Gibbons and Lindenmayer, 2002 p. 120). It therefore seems plausible that the optimal model for selecting habitat trees may need to vary between different forest types. The few studies that have looked at the relationship between forest type and hollow occurrence have been done in areas where the forest types had different dominant tree species (Wormington and Lamb, 1999; Munks *et al.*, in press). However, rates of hollow formation are known to vary between species (Fox *et al.*, 2001; Moloney *et al.*, 2002; Whitford and Williams, 2002). Consequently, the exact nature of the relationship between hollow occurrence and forest type remains uncertain.

Tree hollows in Tasmania

In Tasmania, all eight bat species, the four possums, the sugar glider and approximately 29 bird species use hollows to varying degrees (Chapter 2; Munks *et al.*, in press). All of Tasmania's hollow dependent species have priority status under the Tasmanian Regional Forest Agreement (Commonwealth of Australia and State of Tasmania, 1997), meaning that the distribution and abundance of hollows in Tasmanian forests need to be carefully managed. There is concern about the conservation status of these hollow-using species due to a decline in hollow abundance since European settlement, particularly in 'off-reserve' areas subject to anthropogenic land use practices such as forestry and agriculture. Processes influencing this decline include senescence of retained paddock and urban trees, firewood collection, deforestation as a result of agriculture, plantation development and urban sprawl and production forestry. This decline is continuing despite the fact that over half of Tasmania is classified as forest or woodland, 45.5% of which is in formal reserves and 5.7% of which

is in informal reserves (DPIW, 2001; DPIW, 2006). One result of the Regional Forest Agreement (Commonwealth of Australia and State of Tasmania, 1997) process is that intensification of forestry has occurred in 'off-reserve' forests (Lindenmayer and Franklin, 2002) which is likely to result in a further decline in the hollow resource. The lack of information available on the characteristics and density of hollows required by vertebrate fauna in Tasmania severely hampers the further development of management actions to minimise adverse effects of such habitat disturbance.

In Tasmania's production forests, hollow-bearing trees are currently retained at the landscape level within formal and informal reserves (eg., stream-side reserves, wildlife habitat strips and other areas retained under the Forest Practices system) and at the scale of the harvesting unit (known as a coupe) in patches of trees known as wildlife habitat clumps (Forest Practices Board, 2000). The current requirements for this coupe level of retention outlined in the Forest Practices Code (Forest Practices Board, 2000), are that in areas subject to no or low intensity burning, wildlife habitat clumps should be retained at a rate of approximately 1 clump every 5 ha and should contain a minimum of 2 to 3 habitat trees. In areas exposed to high intensity burning (generally wet forest areas) habitat clumps are to be retained every 200 m along the coupe boundary (Forest Practices Board, 2000). These clumps are suggested to be about 50 m by 20 m in size. A habitat tree is defined as "a mature living tree selected to be retained in a coupe because it has features of special value to wildlife (e.g. hollows)". Habitat trees are selected "on the basis of size and the presence of hollows or the potential to develop hollows over time" (Forest Practices Board, 2000).

The rate of hollow-bearing tree retention in harvesting units is lower in Tasmania than for most other areas of forest in Australia (Wayne *et al.*, 2006). Studies on mainland Australia which have examined the number of trees and hollows that need to be retained in order to perpetuate current animal populations vary from 3 to 30 trees per hectare (Mackowski, 1984: ≥3; Moloney *et al.*, 2002: 5-8; Smith and Murray, 2003: >30). Furthermore, the guidelines provided for habitat tree selection in Tasmania are based on little empirical study and are less specific than almost all other regions of Australia, both in terms of the types of trees to be retained and with regard to variation between forest types. Although recruitment trees are expected to be retained by default in a wildlife habitat clump, no specific guidelines are given to facilitate identification of suitable recruitment trees. "Prescriptions that specify a given number of trees to be retained for cavity-dependent fauna must also account for the inherent variability of forest systems" (Gibbons and Lindenmayer, 1996, p. 255). The Tasmanian protocols do not take into account this variability. A predictive model containing the factors (including site characteristics) influencing the number of hollow-bearing trees and the rate at which trees are being used by fauna will assist the assessment of wildlife habitat

and selection of suitable habitat trees for retention. Creating such models for Tasmanian forests will allow more effective management of forest systems and help to ensure sufficient habitat for the conservation of native vertebrate fauna.

Thesis aims

The aim of this study was to improve substantially the ecological understanding of the abundance and distribution of tree hollows and their use by vertebrate fauna in Tasmania's forests. The paucity of such information in Tasmania prevents appropriate amendments being made to the Forest Practices Code (Forest Practices Board, 2000). Amendments are required in order to better manage the tree hollow resource for hollow-using fauna. The main question to be answered in this thesis is:

How can we predict the availability of usable hollows in the wet and dry *Eucalyptus obliqua* forests of Tasmania?

Eucalyptus obliqua was used as the focal tree species in this study for a number of reasons. Firstly, it is currently and was historically a widely distributed tree species in Tasmania. Secondly, it is the tree species most heavily used by the forest industry ensuring its continued extensive distribution. Thirdly, it is found in both wet and dry forests, allowing a detailed exploration of the influence of site factors on hollow formation and use by fauna. Finally, it has been shown to be a tree species that is heavily used as habitat by fauna (Taylor and Haseler, 1993).

Eucalyptus obliqua is a hardwood species that is widespread in low to medium altitude sites in southeastern Australia (Brooker and Kleinig, 1999). Found in a wide range of forest types, this species can range in form from very tall with a small crown to short with a branching, open canopy (Brooker and Kleinig, 1999). *Eucalyptus obliqua* is absent from poorly drained sites, dry sites and highly infertile soils (Kirkpatrick and Backhouse, 1989).

Thesis structure

This thesis is written as a series of scientific papers.

Chapter 2

"The first step for forest managers is to understand the complexity of the system they are managing and the diverse requirements of the fauna for which they are responsible" (Recher, 1991). Therefore, the first step for this thesis was to investigate which species use hollows in

Tasmania and collate the information available on their distribution, hollow-requirements and conservation status.

Chapter 3

When considering hollow-bearing trees in production forests, one of the main issues of concern is the age at which trees produce suitable hollows. This is because the length of time required for a tree to produce a suitable hollow is generally longer than current harvest rotations (Lindenmayer *et al.*, 1990c; Ball *et al.*, 1999; Gibbons, 1999). Chapter 3 outlines how trees were aged and how the error in these age estimates was calculated.

Chapter 4

This chapter examines how research into the hollow resource should be done and how accurate the results achieved by different methods are likely to be. Two main methods were examined: ground-based surveys and tree-felling surveys. Chapter 4 explores the error associated with both techniques and outlines a way of correcting the number of hollows observed during tree-felling surveys in order to obtain more accurate results.

Chapter 5

Once the background work has been done, one can begin to look at how to predict if a tree will contain a hollow or not. Chapter 5 uses Classification Trees and regression analysis to produce models predicting hollow presence and hollow abundance based on tree and site-level variables.

Chapter 6

Just because a tree has hollows, does not mean it will be used by fauna. Chapter 6 uses Classification Trees and Bayesian regression analysis to model whether a tree or hollow was used by fauna.

Chapter 7

This chapter discusses the implications of the results for our understanding of tree hollows and the management of the hollow resource in Tasmania's production forests.

Sample sizes

The chapters of this thesis largely refer to data collected from trees felled in mature *E. obliqua* logging coupes in State Forest in Tasmania between 2003 and 2005. Sample sizes differ slightly between the chapters. A total of 391 felled trees were examined from 39 sites: 19 dry forest sites, 8 damp and 12 wet. Wood samples for determining tree age (see Chapter

3) were collected from only 329 of these trees at 38 coupes (excluding one dry forest coupe). Wood samples for the other trees could not be obtained due to the nature of the stump after felling and logistical constraints. For five sites (45 trees), no detailed description of the location of hollows was done in the pre-fall survey, meaning that the fate of hollows could not be determined. Consequently these trees were not used in Chapter 4. As a result only 34 sites (15 dry, 8 damp and 11 wet) and 346 trees were used in this chapter. For three trees, no estimate of the proportion of tree unexamined in the post-fall survey was done and so no estimate of corrected hollow abundance could be made. Consequently they were not used in Chapters 5 and 6, giving a sample size of 388 trees in these chapters.

Definitions

For the purpose of this thesis, a hollow is defined as cavity that is at least 2 cm in minimum entrance diameter and hollow depth that is located at least 2 m off the ground. An entrance diameter of 2 cm was chosen as it is believed to be the smallest that is generally used by species. A hollow minimum depth of 2 cm was chosen because it was observed that some species construct nests in shallow scars in the trees. A height of 2 m above the ground was chosen because this was considered sufficient to exclude animals that are not truly arboreal. Hollows are occasionally differentiated into holes and scars, where a scar is a cavity with a smooth, solid back and it appears the tree is growing over the edges of this damaged area. Holes are all other cavities. A hollow-bearing tree is a standing, live tree with at least one hollow. 'Habitat trees' are defined, in this thesis, as live trees retained on a logged site that contain hollows that may be used by fauna. The North American equivalent is 'wildlife trees' although the equivalent for dead trees of 'snags' is frequently referred to in the literature. In this thesis, the term 'hollow-using fauna' refers to vertebrate animals that use tree hollows for nesting and/or roosting. A 'coupe' is a discrete harvesting unit.

Issues not addressed in this thesis

This thesis focuses on the hollow resource, hollow-using vertebrate fauna and the retention of suitable habitat trees in production forest areas in Tasmania. This thesis is concerned only with living trees, although it is acknowledged that dead and fallen trees often provide important habitat. For the purposes of this thesis, a hollow was considered only if it was located at least 2 m off the ground. It is acknowledged that basal hollows can also provide important habitat for fauna, but they were beyond the scope of the current project. Furthermore, this project is not reviewing the adequacy of the tree retention prescriptions that are currently implemented in Tasmania, although the current work has implications for tree retention policy and creates the foundations for further research directed to this end.

Chapter 2

The hollow-using vertebrate fauna of Tasmania – distribution, hollow requirements and conservation status

Abstract

In Tasmania, a considerable proportion of the forested landscape is available to land clearance and production forestry (54.4% of forested areas), which has and will continue to result in a substantial decrease in hollow availability. Evidence of the concern for native vertebrate fauna as a result of these activities is the fact that all hollow-using species are listed as having priority status under the Tasmanian Regional Forest Agreement. This status means that a commitment has been made to ensure careful management of these fauna. In order to ensure the habitat for hollow-using fauna is managed effectively, we first have to understand the requirements of the species involved. This paper is a review of the distribution, hollow requirements and conservation status of the five arboreal marsupials, eight bats and 29 birds that use hollows in Tasmania. The number of species which use hollows is lower than many other areas of Australia but they represent a large proportion of the vertebrate fauna of Tasmania (31.9% of terrestrial birds and 35% of terrestrial mammals). Three of these species and nine sub-species are endemic to Tasmania and four bird species are listed as threatened at the state and/or national level. Twenty-five of Tasmania's hollow-using species are capable of using small hollows, 14 can use medium-sized hollows while three bird species are limited to using large hollows. Although distributed across the state, current records indicate that the highest diversity of hollow-using species is found on the north coast and southeast corner of Tasmania, in association with dry forest areas.

Introduction

A tree hollow is a naturally-formed cavity in the live or dead wood of a tree. Tree hollows provide important roosting and nesting habitat for many animals worldwide (Maheswaran and Balasubramanian, 2003; Martin *et al.*, 2004; Kalcounis-Ruppell *et al.*, 2005; Ruczynski and Bogdanowicz, 2005; Walker *et al.*, 2005; Monterrubio-Rico and Escalante-Pliego, 2006). Australia has a large number of hollow-using species, but has no primary excavators such as woodpeckers. Although some vertebrate species in Australia are known to modify the size of cavities, for example, cockatoos (Saunders *et al.*, 1982), hollows are usually

produced by fire, fungi and/or termites. Consequently, many years are required to form hollows, particularly large hollows, in Australian forests. There is a general decline in hollow-bearing trees across Australia, due to land clearing for urbanization and agriculture, forestry activities and the dying out of hollow-bearing trees retained in paddocks and urban areas (Gibbons and Lindenmayer, 2002). This has resulted in concern for hollow-using fauna across the country (Gibbons *et al.*, 2000b; Whitford and Williams, 2002; Wormington *et al.*, 2002).

A review by Munks *et al.* (in press) found that 45 vertebrate species have been recorded using tree hollows for nesting or shelter in Tasmania. These species include endemics (at the species or subspecies level) and species listed in the schedules of the Tasmanian *Threatened Species Protection Act 1995* and the Commonwealth *Environmental Protection and Biodiversity Conservation Act 1999.* Approximately 54% of Tasmania (3.67 million hectares) is forested and so potentially provides habitat for such species. Of this forested area, 45.6% (1.67 million ha) is reserved (1.46 million ha in formal reserves and 0.21 million ha in informal reserves) (DPIW, 2001; DPIW, 2006). However, there is unequal reservation of forest types both by community and location. Of Tasmania's 50 forest communities, only thirty-four meet the CAR (Comprehensive, Adequate and Representative) reserve (ANZECC and MCFFA, 1997) requirements by having at least 15 % of their pre-1750 extent protected in reserves. Ten forest communities have less than 7.5% of their pre-1750 extent in reserves (Resource Planning and Development Commission 2002, cited in Hickey and Brown, 2003). In addition, little forest is reserved in the southeast of the state where human populations are higher than other areas of Tasmania (Hickey and Brown, 2003). Of the 1.997 million ha of forest that is unreserved, 1.002 million ha is State Forest and 0.995 million ha is private land (DPIW, 2001; DPIW, 2006). The large majority of this forest is potentially available for timber harvesting and is therefore unlikely to provide habitat for hollow-using fauna into the future unless managed appropriately. It is particularly important that habitat for hollow-using fauna is conserved in areas of the state where forest is poorly reserved and in those forest types that are not adequately captured within the CAR reserve system. Without adequate conservation measures the future status of the hollow resource and its associated fauna in Tasmania is uncertain.

Forest management agencies in most states of Australia, including Tasmania, have developed management prescriptions for the conservation of habitat for hollow-using fauna in 'off-reserve' areas (Wayne *et al.*, 2006). Studies have shown that the hollow requirements and the degree to which fauna are dependent on hollows vary greatly between species. Consequently, knowledge of the fauna that use hollows in a particular region and their known or likely hollow requirements is essential for the development of effective

management actions (Recher, 1991). Tasmania's Regional Forest Agreement (Commonwealth of Australia and State of Tasmania, 1997) recognises hollow-dependent fauna as a priority fauna group to be protected by actions prescribed in the Tasmanian Forest Practices Code (Forest Practices Board, 2000). In the absence of information on the hollow resource and its associated fauna in Tasmania, the current Tasmanian management prescriptions were developed predominantly from studies conducted on mainland Australia (Taylor, 1991). To assist in the revision of these current measures for the conservation of habitat for hollow–using fauna in Tasmanian production forests this paper reviews current knowledge (published and unpublished) on the hollow requirements, distribution, threats and status of vertebrate species known to use tree hollows in Tasmania. This synthesis allows identification of the breadth of species requirements and identifies knowledge gaps that require further research. This review is not intended to be a complete account of the requirements of hollow-using fauna in Tasmania. It does not include information on reptiles or amphibians that may use hollows or the wide range of invertebrate species that are likely to use hollows as hibernation and aestivation sites. The aim of this chapter is to summarize information on the distribution, hollow requirements and conservation status of the main vertebrate species which use hollows (arboreal marsupials, bats and birds) for breeding and/or refuge.

Sources of information

Information for this review was obtained from a variety of sources, including published literature (largely from Tasmania but also mainland Australia where indicated), unpublished reports, a community-based survey (see Appendix 1), examination of museum specimens and personal communication with experts.

Museum specimens were examined where available in order to provide a rough indication of the minimum size of hollow required by the species in Tasmania. Classification of animal species into different hollow size categories was done according to the size of the head of the animal and then verified by experts. Hollow sizes required by the species covered in this review were categorized into small (2-5 cm entrance width), medium (6-10 cm entrance width) and large (>10 cm entrance width). These classifications represent minimum hollow entrance dimensions that can be used by an animal. They do not reflect the internal dimensions required by fauna and many species will use hollows with entrances of a larger size. Small, medium and large hollow users will be addressed separately in this chapter. Initially, the group of animals in a particular size class will be considered as a whole and then each species within that size class will be addressed separately. Due to the small size and cryptic nature of the bats found in Tasmania, many studies do not differentiate between

species. Consequently, bats will first be addressed as a group and then any relevant species-specific information that is available will be presented.

There were large differences in the amount of literature available between species, with more information generally available for threatened species. Due to the limited amount of information that was available for some species of birds in Tasmania, a questionnaire survey was distributed to Tasmanian ornithologists (amateur and professional) (see Appendix 1). The aim of this survey was to gain opinions on the degree to which a particular bird species uses hollows, whether the population size of the species is increasing, stable or decreasing and any threatening processes. Birds considered were those previously reported to use tree hollows in Tasmania (Munks *et al.*, in press) and any others that might use hollows. A major source of information on the distribution of birds was the bird atlases (Thomas, 1979; Blakers *et al.*, 1984; Barrett *et al.*, 2003). Differences in population trends detected in the two national Atlas' (Blakers *et al.*, 1984; Barrett *et al.*, 2003) are subject to error resulting from a difference in survey methods used. Attempts were made by the authors of these books to correct this error (Barrett *et al.*, 2002), but caution should be taken in interpretation and the Atlas' should be consulted directly for more information.

For the distribution maps produced in this paper, the majority of bird sightings were provided by Birds Australia. Information on bat distributions was obtained from Rounsevell *et al.* (1991), who obtained their distribution data from the Tasmanian Parks and Wildlife Service 'TASPAWS' database. Arboreal mammal distribution data was obtained from Rounsevell *et al.* (1991), Munks *et al.* (2004a) and from the Department of Primary Industries and Water Natural Values Atlas database (DPIW, 2007). In addition, several experts were approached to provide further unreported animal sightings. Data from these sources was often collected on an opportunistic basis post 1964 and is therefore likely to be concentrated around more populated areas or areas with well-used roads. The data represents animal sightings and not just nest sightings and so the presence of some species may not necessarily reflect hollow use in an area. This is most likely to have occurred with the yellow-tailed black cockatoo which is a highly visible and mobile species. For example, around Hobart it is only known to nest on Mount Wellington but sightings have been reported in surrounding areas (M. Holdsworth pers. comm.). Many records on King Island are also unlikely to reflect patterns of hollow use due to a lack of hollow-bearing trees.

Results

This review identified 42 species that use hollows for breeding and/or refuge in Tasmania; five marsupials, eight bats and twenty-nine birds. Twenty-five of these species were

classified as small hollow users, 14 as medium hollow users and three as species that are only able to use large hollows (Tables 2.2, 2.3 and 2.4). Species reported as being 'hollow users' in other sources but which are not included in this review are listed in Table 2.1. Vagrant species that have only been sighted a few times in Tasmania and species known to use tree hollows only very occasionally will not be considered in the current paper. Some species that are known to use tree hollows on mainland Australia are not believed to do so in Tasmania. Species that only use hollows found at the base of a tree (basal hollows) are also not considered. Basal hollows are generally not included in studies of tree hollows (e.g. Gibbons *et al.*, 2000b), presumably because the type of habitat they provide is readily exchangeable for a hollow log or other refuge site on the ground. The remainder of this chapter will focus on the main hollow-using vertebrate fauna currently present in Tasmania.

Small hollow users

Twenty-five species were identified as being able to use small hollows in Tasmania (Table 2.2). This included 14 bird species, three marsupials and eight bats. Locality records for small hollow users are prevalent across the state, but highest on the north coast and in the southeast, with fewer species in the west (Figure 2.1). This roughly corresponds with forest type, the western third of the state having more rainforest and wet sclerophyll forest in comparison to the rest of the state which is predominantly dry sclerophyll forest and non-forest areas. Details on the specific habitats used by each of the species are provided in Table 2.2. For the few species where information on the hollow requirements in Tasmania was available, hollows may be as close together as one metre, or more than a kilometre apart. Up to 20 nests/ha may be used by one species (Table 2.2). Three of the small hollow users have been classified as endangered (Table 2.2). Two species which use small hollows are endemic and so found only in Tasmania while a further four are endemic at the subspecies level. Six species are migratory, over-wintering on mainland Australia and then returning to Tasmania to breed.

Table 2.1 A summary of Tasmanian species which have been classified as hollow-using by other sources but which will not be addressed in this text and the reason for their exclusion.

Species	Reason not included*
Gang gang cockatoo (*Callocephalon fimbriatum*)	Vagrant (rarely sighted in Tasmania)
Sacred kingfisher (*Halcyon sancta*)	Vagrant (rarely sighted in Tasmania)
Azure kingfisher (*Alcedo azurea*)	No evidence of hollow-use in Tasmania
Black currawong (*Strepera fuliginosa*)	Rarely uses hollows (but does occasionally, C. Spencer pers. comm.)
Scarlet robin (*Petroica multicolor*)	Rarely uses hollows (hollow use declared by Haseler, 1990)
Spotted pardalote (*Pardalotus punctatus*)	Rarely uses hollows
Grey shrike-thrush (*Colluricincla harmonica*)	Rarely uses hollows (hollow use declared by Sharland, 1958; Green, 1995)
Satin flycatcher (*Myiagra cyanoleuca*)	Rarely uses hollows (hollow use declared by Statham, 1987)
Tawny frogmouth (*Podargus strigoides*)	Rarely uses hollows (hollow use declared by Statham, 1984)
Barn owl (*Tyto alba*)	Vagrant (rarely sighted in Tasmania)
Brown falcon (*Falco berigora*)	Does not use hollows in Tasmania (although there is one anecdotal report of hollow use: C. Spencer pers. comm.)
House sparrow (*Passer domesticus*)	Rarely uses hollows (hollow use declared by Green, 1995)
Spotted-tailed quoll (*Dasyurus maculatus*)	Rarely uses hollows (hollow use declared by Munks et al., in press)
Echidna (*Tachyglossus aculeatus*)	Rarely uses hollows (but see Wapstra *et al.*, 2000)
Dusky antechinus (*Antechinus swainsonii*)	Hollow use little known but thought to use basal hollows or hollow logs (declared by Statham, 1984)
Swamp antechinus (*Antechinus minimus*)	Hollow use little known but thought to use basal hollows or hollow logs (declared by Statham, 1984)
Pretty skink (*Niveoscincus prettiosum*)	Little is known about hollow use by this species but it is assumed to be rare (hollow use declared by Munks et al., in press)
Metallic skink (*Niveoscincus metallicus*)	Rarely uses hollows (hollow use declared by Munks et al., in press)
Brown tree frog (*Litoria ewingii*)	Rarely uses hollows (hollow use declared by Munks et al., in press)

* Where it is referred to that hollow use is 'declared' by someone, the reference does not provide evidence of hollow use but a comment is made by the author that they use hollows.

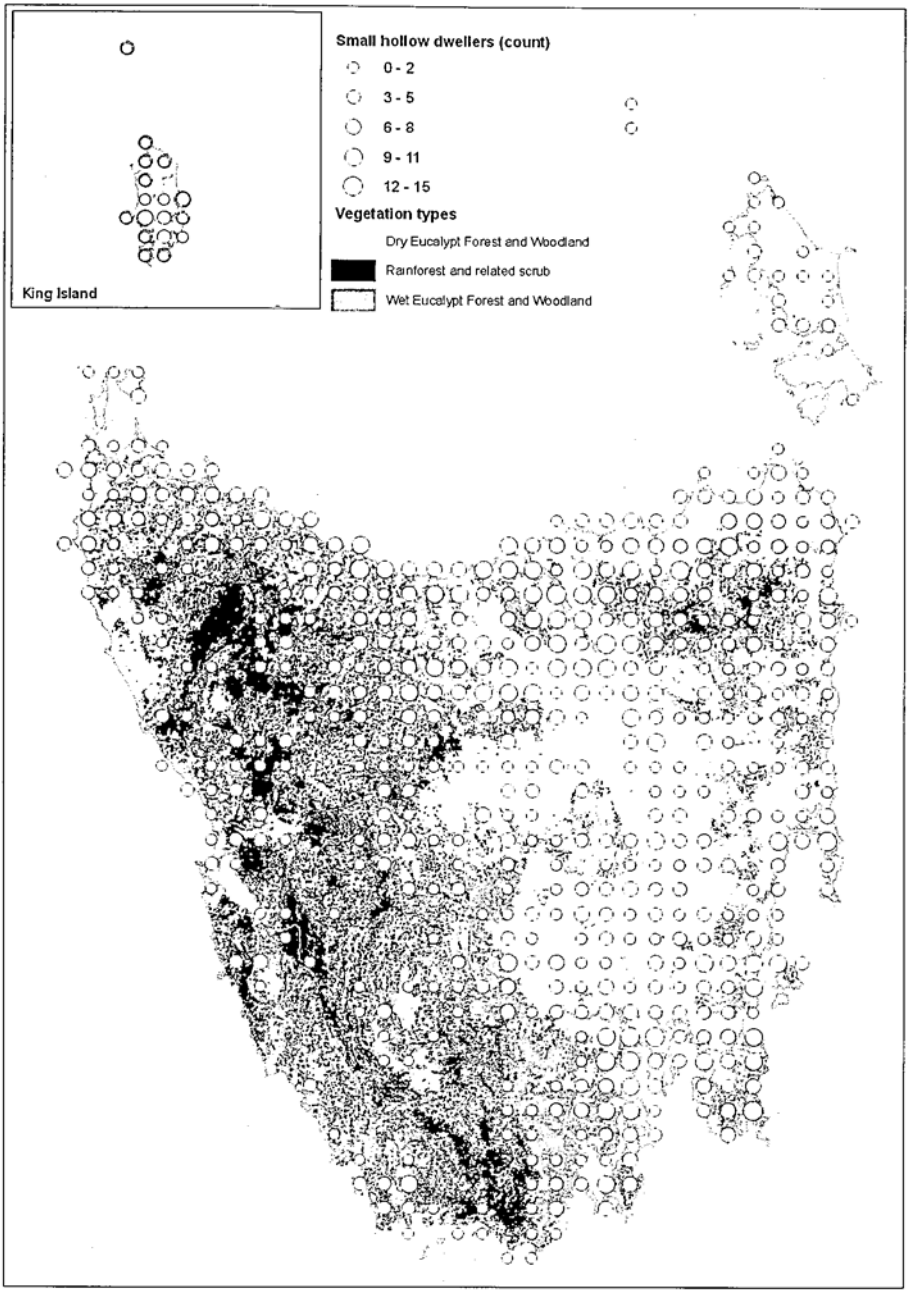


Figure 2.1 Map indicating the number of species that are able to use small hollows that are known to exist in 10 x 10 km grid squares around Tasmania. Small hollows have a minimum entrance width of 2-5 cm. Indication is also given of the type of forest found in the different areas of the state.

Table 2.2 Conservation status, hollow use, habitat, population density, density and spacing of nest trees for animals that use small hollows (2-5 cm entrance width) in Tasmania. Blank cells indicate no information was available.

Species	Status	Hollow use	Habitat type	Animal density (animals/ ha)	Number nest trees used per ha	Calculated spacing of nest trees (m)	References
Musk lorikeet	t	B G, (S)	D, SW, A, U	0.14 (T)		Can nest in same tree (M)	Lord, 1924; Cayley and Lendon, 1973; Thomas, 1979; Green, 1995; Taylor *et al.*, 1997
Rainbow lorikeet	-	B G, (S)	A			Can nest in same tree (M)	Anonymous, 1919; Fielding, 1979
Swift parrot	Ee	B G, (S)	D, CH, SW, W, A, Wd, U	0.02 – 0.83 (T)	0.04 – 0.5 (T) 0.7-0.8 nests /ha (T)	Two or more can be in same tree (T)	Sharland, 1958; Recher *et al.*, 1971; Bosworth *et al.*, 1976; Thomas, 1979; Wilson, 1984; Brown, 1989a; Taylor *et al.*, 1997; Webb *et al.*, 2007
Orange-bellied parrot	MEe	B S, G	S, R, C, M, A, Wd			Two or more can be in same tree (T)	Thomas, 1979; Brown and Wilson, 1984; Green, 1995; Orange-Bellied Parrot Recovery Team, 1999; Holdsworth, 2006
Blue-winged parrot	M	B S, G	S, M, Re, Wd, (CH, SW, D, A)	0.1 – 0.62 (T)		Nests are often grouped together (T)	Giblin and Swindells, 1927; Recher *et al.*, 1971; Bosworth *et al.*, 1976; Green, 1977; Thomas, 1979; Green, 1995; Taylor *et al.*, 1997
Tree martin	M	F G, (C, S)	D, Wd, A, M, CH, W, MF, S, Re, U	0.14 – 1.11 (T) 0.04-0.23 territories/ha (T)		> 0.33 and many can occur in the same tree (M)	Slater, 1962; Recher *et al.*, 1971; Bosworth *et al.*, 1976; Green, 1977; Bell, 1979; Thomas, 1979; Loyn, 1980; Green, 1995; Taylor *et al.*, 1997
Welcome swallow	M	R S, (G)	A, CH, SW, ML, Re, U, (MF, S, M)	0.25 – 0.88 (T)		< 200 (T) 1-14 (M)	Sharland, 1958; Bosworth *et al.*, 1976; Green, 1977; Thomas, 1979; Park, 1981; Tarburton, 1993; Taylor *et al.*, 1997
Australian owlet-nightjar	t	BS M	D, Wd, (W)			Loosely colonial spaced by territories (< 80 ha) (T) Can nest close together (M)	Thomas, 1979; Hollands pers. comm. in Higgins, 1999; Bryant, 2002

Table 2.2 continued

Species	Status	Hollow use	Habitat type	Animal density (animals/ ha)	Number nest trees used per ha	Calculated spacing of nest trees (m)	References
Dusky robin	T	R S	CH, D, W, SA, S, R, P, CF, ML, A, (MF)	0.01-1.39 (T)			Fielding, 1976; Fielding, 1979; Thomas, 1979; Coulson and Coulsen, 1981; Wilson, 1984; Taylor et al., 1997
Flame robin	-	R S	CH, Wd, D, W, SA, A, M, S, R, (MF, CF)	0.19-1.07 (T) 0.04-0.59 territories/ha (M)	19.8 nests/ha (T) 0.044 nests/ha (M)	131±54 (M)	Fletcher, 1924; Elliott, 1931; Thomas, 1979; Loyn, 1980; Coulson and Coulsen, 1981; Robinson, 1990; Taylor *et al.*, 1997
Forty-spotted pardalote	TEe	B G, (S)	D, Wd, C, (W)	0.68-4.95 (T)	0.44-8 (T)	> 5 (T)	Thomas, 1979; Statham, 1984; Woinarski and Bulman, 1985; Brown, 1986; Brown, 1989b; Bryant, 1997
Striated pardalote	M	B G, (S)	D, CH, SW, W, MF, R, SA, M, Wd, Re, A	0.65 – 2.59 (T) 0.32 breeding pairs/ha (T) 30 pairs in 90 m^2(T)	0.27 (T)	> 2 (T) Nest in loose colonies (T)	Sharland, 1958; Recher *et al.*, 1971; Green, 1977; Thomas, 1979; Woinarski and Bulman, 1985; Taylor and Haseler, 1993; Green, 1995; Taylor *et al.*, 1997
Dusky woodswallow	M	F S, (G)	CH, SW, D, S	0.24 – 1.19 (T)		As close as a few metres (M)	Sharland, 1958; Thomas, 1979; Coulson and Coulsen, 1981; Taylor *et al.*, 1997; Higgins *et al.*, 2006
Common starling	-	F G, (S)	CH, SW, D, Wd, A, U	0.02 – 0.12 (T) 0.44-2.30 territories /ha (M)		In same tree or up to 800 m away (M)	Thomas, 1957; Recher *et al.*, 1971; Bosworth *et al.*, 1976; Thomas, 1979; Lenz, 1990; Green, 1995; Taylor *et al.*, 1997
Sugar glider	-	F C	R, W, D, A	0.09-12 (M)	0.14-10 (M)	32-100 (M)	Smith and Lees, 1998; Bryant, 2002; Gibbons and Lindenmayer, 2002; Smith and Murray, 2003; Munks *et al.*, 2004a
Eastern pygmy possum	t	F C	R, W, D, Wd, S, U, M, A		18 nests /ha (T) 5-45 (M)	25-133 (T) 15-45 (M)	Hocking and Guiler, 1983; Ward, 1990; Duncan, 1995; Duncan and Taylor, 2001; Gibbons and Lindenmayer, 2002; Munks *et al.*, 2004a

Table 2.2 continued

Species	Status	Hollow use	Habitat type	Animal density (animals/ ha)	Number nest trees used per ha	Calculated spacing of nest trees (m)	References
Little pygmy possum	-	F C	D, Wd, W, SA, A, S			25-88 (T)	Duncan, 1995; Duncan and Taylor, 2001; Munks *et al.*, 2004a
Lesser long-eared bat	-	B C (M)	U, D, Re, W, R, C, ML		1-3 per individual unknown area (T) 1-8 per individual unknown area (M)	80-1400 unspecified species (T)[a]	O'Neill, 1984; Taylor et al., 1985; O'Neill and Taylor, 1986; Taylor et al., 1987; Lumsden et al., 2002
Greater long-eared bat	t	B C (M)	W, C, ML				O'Neill, 1984; O'Neill and Taylor, 1986; Taylor *et al.*, 1987
Gould's wattled bat	-	B C (M)	W, Wd, A		1-6 per individual unknown area (M)		O'Neill, 1984; O'Neill and Taylor, 1986; O'Neill and Taylor, 1989, Lumsden *et al.*, 2002
Chocolate wattled bat	-	B C (M)	W, C, Re, D, R, ML		1-2 per individual unknown area (T)	80-1400 unspecified species (T)[a]	O'Neill, 1984; Taylor *et al* , 1985; O'Neill and Taylor, 1986
Little forest eptesicus	-	B C (M)	D, C, Re, ML			3.6 (T)	Green, 1965; O'Neill, 1984; O'Neill and Taylor, 1986
King river vespadelus	-	B C (M)	C, Re, D, W, R, ML		1-3 per individual unknown area (T)	80-1400 unspecified species (T)[a]	O'Neill, 1984; Taylor *et al.*, 1985; O'Neill and Taylor, 1986
Large forest vespadelus	-	B C (M)	D, Re, W, R, C, ML		1-3 per individual unknown area (T)	80-1400 unspecified species (T)[a]	O'Neill, 1984; Taylor *et al.*, 1985, O'Neill and Taylor, 1986
Tasmanian pipistrelle	-	B C (M)	W, C, Re, U, ML, D				Green, 1977; O'Neill, 1984; O'Neill and Taylor, 1986

STATUS: M = migratory, T = Tasmanian endemic, t = Tasmanian subspecies is endemic (although sources differ in their decision on this), E and e = endangered under Commonwealth *Environmental Protection and Biodiversity Conservation Act 1999* (EPBC) and Tasmanian *Threatened Species Protection Act 1995* (TSPA), respectively. V and v = vulnerable under EPBC and TSPA, respectively. R and r = rare under EPBC and TSPA, respectively.

HOLLOW USE: *Degree of dependency*: BS = requires hollows for breeding and shelter (i.e. roosting or denning), B = breeding, requires hollows for breeding, F = facultative, uses hollows when available but can use other breeding and/or nesting sites, R = rarely, rarely uses hollows. (Note, even for the breeding-shelter and breeding categories, there have often been observations of nesting in non-hollows, but if the vast majority of instances are in hollows they are considered to be eligible for these categories). *Sociality of nesting*: G = gregarious, many pairs nest in close proximity, each requiring one hollow, S = single, each pair requires only one hollow which may be isolated from other members of the species, C = communal, hollow shared by more than one individual but each individual uses multiple hollows, M = multiple, multiple hollows are used by each individual but nesting is usually solitary. Brackets indicate this is done only occasionally.

HABITAT TYPE: R = rainforest, W = wet forest, D = dry forest, MF = mixed forest, Wd = woodland, CF = coniferous forest, ML = mallee, CH = coastal heath, C = coastal, S = sedgeland, SW = savannah woodland, M = moorland, Al = alpine, SA = subalpine, A = agricultural land, P = plantation, Re = regrowth, U = urban, Fr = freshwater, Td = tidal. Brackets indicate low densities.

CALCULATED SPACING OF NEST TREES: These figures represent the distances apart used nests have been found. For species that use multiple nests it may refer to distances between nests used by an individual. For species that only require one hollow per breeding pair, it will be the distance between nests of different individuals.

(M) indicates results are from mainland Australia, (T) indicates results are from Tasmania.

[a] These distances were indicated in a study examining these four species but it was not specified which exact species the spacing referred to.

Musk lorikeet (*Glosopsitta concinna didimus*)

Distribution: This species is largely found in the drier eastern half of the state (Thomas, 1979), although some unconfirmed records have been received from the southwest where it may be a vagrant (Brown, 1979). It is thought to avoid areas of high rainfall and high elevation (Brown, 1979).

Hollow requirements: Musk lorikeets are reliant on hollows for breeding but not roosting (Appendix 1), nesting in both trees and stumps (Sharland, 1958). Little is known about their specific hollow requirements. Three described nests were found between 5.5 and 12.2 m up a tree and between 38 to 61 cm down the hollow (Australian National Wildlife Collection).

Status and threats: The status of the musk lorikeet is unclear. Six out of fourteen responses to the survey indicated that the species is declining, while the 'expert' was the single response saying the species is increasing (Appendix 1). In Bryant (2002) they are stated as being declining while the Atlas data suggests they may be increasing across Australia (although they were also more likely to be recorded in the second survey: Barrett *et al.*, 2003). This indicates further research is warranted. Although forestry is seen as a major threat to this species (Appendix 1) they do not generally use forested areas (M. Holdsworth pers. comm.). The main threats to the species are considered to be clearing for agriculture, loss of habitat due to firewood collection and competition for nest sites. Concern has also been expressed over poaching, road kill and injury by window collision (Appendix 1; Bryant, 2002).

Rainbow lorikeet (*Trichoglossus haematodus*)

Distribution: Sightings have been received from the Ulverstone and Hobart areas (Birds Australia Atlas data) although they are likely to be found in a range of areas.

Hollow requirements: Rainbow lorikeets use tree hollows for breeding but not roosting (Appendix 1; Sharland, 1958).

Status and threats: They are rarely sighted and are possibly vagrants in Tasmania, although reported sightings go back many years (Cooper, 1998). Numbers of this species are believed to be stable or increasing (Appendix 1) and this species is now established in Tasmania (M. Holdsworth pers. comm.). Due to their dubious native status and rare nature, threats are not considered although this species may provide extra competition for native hollow-using fauna.

Swift parrot (*Lathamus discolor*)

Distribution: The swift parrot breeds in Tasmania and migrates to mainland Australia in autumn. In Tasmania they are mainly found in the east, although there are records from across the state (Brown, 1979; Thomas, 1979). The main breeding range is the southeast

coast, but the species is known to breed in the northwest (Brereton, 1997b; Brereton *et al.*, 1999; Webb *et al.*, 2007) and northeast coast (M. Webb unpublished data). Nesting sites are generally in relatively close proximity to the main food source (*E. globulus* nectar) and near large forest patches with old growth characteristics (Webb *et al.*, 2007).

Hollow requirements: Swift parrots require tree hollows for breeding but not roosting (Appendix 1; Sharland, 1958; Brown, 1989a), often using large *E. obliqua, E. pulchella* and *E. globulus* trees (Brereton, 1997b). Most nests observed are in hollows in branches (Brereton, 1997b; Webb *et al.*, 2007). The height of the hollow above the ground varies from 5 to 40 m (most between 6 and 20 m) in trees an average of 105 cm in diameter (range 33 to 305 cm) (Brereton, 1997b; Webb *et al.*, 2007). Brereton (1997b) found that hollows generally had a mean entrance diameter of 10 ± 4 cm (Brereton, 1997a) while Webb *et al.* (2007) estimated that the entrances were usually smaller than about 6 cm in entrance diameter. The depth of hollows used ranges between 30 to 60 cm (Australian National Wildlife Collection). They are generally a communal nesting species, often found in close proximity, even in the same tree (Brown, 1989a; Brereton, 1997b; Brereton *et al.*, 1999; Webb *et al.*, 2007).

Status and threats: This species is listed as endangered in the state and nationally (Commonwealth *Environmental Protection and Biodiversity Conservation Act* 1999, Schedule 3 Tasmanian *Threatened Species Protection Act* 1995) due to loss of habitat and the small and decreasing population size (Brereton *et al.*, 1999; Swift Parrot Recovery Team, 2001). Previously common, it was noted to be declining in the early 1980s (Wilson, 1984). Population size estimates dropped from 1320 pairs in 1987 (Brown, 1989a) to 940 pairs in 1995 (Swift Parrot Recovery Team, 2001). There are currently estimated to be less than 1000 pairs or 2500 mature individuals (Brereton *et al.*, 1999; Swift Parrot Recovery Team, 2001). A recovery program was established in 1995 (Brereton *et al.*, 1999). The main threat to this species is loss of habitat through forestry operations, agricultural clearing, urbanization and firewood collection (Brereton *et al.*, 1999). Current figures are that of the 8000 ha of foraging habitat in the breeding range, 82% is on private land and only 18% is in CAR reserves (Swift Parrot Recovery Team, 2001). While foraging habitat has generally received a high level of protection, recent studies have indicated little protection is provided for nesting hollows throughout the range (Webb *et al.*, 2007). Further pressure on this species occurs from competition for nest hollows by starlings, tree martins, owlet nightjars, green rosellas and blue-winged parrots (M. Webb pers. comm.; Wilson, 1984; Brown, 1989a), collision with cars, fences and windows (Brereton *et al.*, 1999; Swift Parrot Recovery Team, 2001).

Orange-bellied parrot (*Neophema chrysogaster*)

Distribution: The orange-bellied parrot migrates to mainland Australia (Victoria and South Australia) in winter and returns to Tasmania to breed (Brown, 1979; Thomas, 1979; Drechsler, 1998). They are found patchily across the north and west coast, with a few records in the southeast and central northeast. Although previously reported to breed in the Midlands of Tasmania (Giblin and Swindells, 1927; Sharland, 1947; Brown and Wilson, 1982), these may be misidentified reports of the blue-winged parrot (M. Holdsworth pers. comm.). Current evidence indicates that this species only breeds in the southwest close to the coast or large harbours (Brown, 1979; Thomas, 1979; Statham, 1984; Holdsworth, 2006). Most of the breeding population is located within the Bathurst Harbour area, centred on Melaleuca (Holdsworth, 2006).

Hollow requirements: Orange-bellied parrots rely on tree hollows for breeding (Appendix 1), using either isolated copses of trees or trees located in large areas of forest (Brown and Wilson, 1982). They generally roost in thick heath along creeks (Brown and Wilson, 1982). Nesting hollows are generally located 8–25 m up live *Eucalyptus nitida* or *E. ovata* trees. The nest entrance is usually about 5 cm across (but up to 25 cm) and 45-60 cm deep (Brown and Wilson, 1982; Brown and Wilson, 1984).

Status and threats: The orange-bellied parrot is listed as endangered both in the state and nationally (Commonwealth *Environmental Protection and Biodiversity Conservation Act* 1999, Schedule 3 Tasmanian *Threatened Species Protection Act* 1995) due to a decrease in numbers and range since European settlement (Lord, 1924; Brown and Wilson, 1982). Species recovery efforts are occurring and include population monitoring, habitat management and captive breeding (Orange-bellied Parrot Recovery Team, 2006). The total population is believed to be around 150 mature adults (Orange-bellied Parrot Recovery Team, 2006). Although reasons for the decline and factors influencing the mortality of the species are unclear (Drechsler, 1998), the main threat is presumed to be loss of wintering habitat. Other potential contributing factors are changes in fire regime in the breeding habitat, competition for food, predation, disease, windfarms and possibly inbreeding depression (Brown and Wilson, 1984; Orange-Bellied Parrot Recovery Team, 1999).

Blue-winged parrot (*Neophema chrysostoma*)

Distribution: Although records of the blue-winged parrot are mostly centred in the Midlands region, they are widely distributed across the state (Giblin and Swindells, 1927; Thomas, 1979). During the breeding season they are usually found in open forests with hollow-bearing trees (Giblin and Swindells, 1927; Sharland, 1947). The majority of individuals migrate to the mainland in winter (Brown, 1979).

Hollow requirements: Blue-winged parrots are dependent on tree hollows for breeding but not roosting (Appendix 1; Giblin and Swindells, 1927). Some reports indicate they often breed in retained stags in cleared forest (Wilson, 1981). The preferred nesting location is a dead spout with a hard outer casing and a decaying centre or a broken off upright branch. They have also been known to use knot-holes in the upper side of a horizontal branch and occasionally a hole in the main bole of the tree. They like trees with hollow spouts or decaying branches (Giblin and Swindells, 1927). Nests have been located between 3 and 24 m from the ground (Australian National Wildlife Collection; Giblin and Swindells, 1927). The depth of the hollows used ranges from 30-60 cm (Australian National Wildlife Collection; Giblin and Swindells, 1927). Individuals have been found nesting within several feet of each other in the same tree (Giblin and Swindells, 1927).

Status and threats: Reports on the current status of this species are conflicting. Some reports indicate an increase in numbers and range due to land clearing (Giblin and Swindells, 1927; Brown, 1979; Brown and Wilson, 1982) while others report a decline since European settlement (Lord, 1924; Green, 1983) that may be continuing today (Appendix 1). The Atlas data indicates there has been little change in population size across Australia (Barrett *et al.*, 2003). The potential increase in range causes some concern as it may start expanding further into the range of the endangered orange-bellied parrot (Brown and Wilson, 1982). The main threats to this species are considered to be forestry, clearing for agriculture, competition for nest hollows, windfarms and poaching (Appendix 1; Giblin and Swindells, 1927). However, they may be found in regenerating forest if nest trees are available (Pattemore, 1980).

Tree martin (*Hirundo nigricans nigricans*)

Distribution: Tree martins are widespread in the east of the state, being more scattered in the west and north (Thomas, 1979). They are frequently observed in populated areas (Thomas, 1972; Fielding, 1979; Harris, 1980) and are characteristic of open environments (Taylor *et al.*, 1997). Some reports suggest they breed only in mature forest (Wilson, 1981) although no difference in density was found between mature forest and regrowth by Taylor *et al.* (1997).

Hollow requirements: Tree martins mostly use tree hollows for nesting but can use other sites such as caves and holes in cliffs (Australian National Wildlife Collection) and buildings (Sharland, 1958; Harris, 1980). They use hollows with small entrances (Green, 1995) and have been found 4.5 and 12 metres above the ground (Dove, 1916).

Status and threats: This species is believed to be stable at moderate numbers although there is some concern of a decline (Appendix 1). The Atlas data indicates a potential decrease over most of Tasmania except in the central and southern areas (Barrett *et al.*, 2003). The major threats are perceived to be the forest industry, clearing for agriculture and competition for

nests (Appendix 1). However, Wapstra and Taylor (1998) only observed tree martins after logging. Concern has also been expressed about wind farms (Appendix 1).

Welcome swallow (*Hirundo neoxena*)

Distribution: Welcome swallows are found across the state, including occasionally in the southwest (Thomas, 1979). This species is characteristic of open environments (Ratkowsky and Ratkowsky, 1977; Taylor *et al.*, 1997) and is frequently seen in populated areas (Thomas, 1972; Fielding, 1979; Harris, 1980) although it has also been recorded near wet sclerophyll forests (Ratkowsky and Ratkowsky, 1977).

Hollow requirements: It is likely that this species only occasionally uses tree hollows for nesting (Appendix 1; Sharland, 1958). It will often build nests around man-made structures (Sharland, 1958; Green, 1977; Park, 1981). Of the eleven nests found by Green, six were in man made structures, three were in hollows 1.5-2.7 metres up trees, one was under a sea coast cliff ledge and the other location was unclear (Australian National Wildlife Collection). It is believed that welcome swallows favour large trees where they nest in burnt out trunks with openings facing north or east, preferring hollows with more than one entrance (Fletcher, 1924).

Status and threats: Populations are thought to be stable at relatively high numbers (Appendix 1). They are thought to benefit from logging (Green, 1977; Pattemore, 1980; Taylor *et al.*, 1997), although Wilson (1981) states that while they feed over regenerating forest they nest in hollow and burnt older trunks. Welcome swallows are not perceived to be under any major threat.

Australian owlet nightjar (*Aegotheles cristatus*)

Distribution: Australian owlet nightjars are widely distributed in the east of Tasmania (Thomas, 1979) although they have recently been found breeding in the southwest of the state (Orange-Bellied Parrot Recovery Team, 1999).

Hollow requirements: Australian owlet nightjars are reliant on hollows for nesting and roosting, although there was a report of roosting in a fence post (Appendix 1; Sharland, 1958; Statham, 1984; Bryant, 2002). Green located three nests (Australian National Wildlife Collection), all in hollows which were located between 4.5 and 5.5 metres up the trees. The hollows used were around 75 cm deep (Australian National Wildlife Collection). Colonies are loosely spaced by territories which tend to be less than 80 ha in size (Bryant, 2002).

Status and threats: There is some concern that this species is found only in low numbers or declining (Appendix 1). The greatest threats are perceived to be competition for nest sites, clearing for agriculture and forestry and road kill (Appendix 1).

Dusky robin (*Melanodryas vittata*)

Distribution: This Tasmanian endemic is common but rarely abundant (Wilson, 1984) and is widely distributed around the state (Thomas, 1979). While probably more numerous in open forest, they are found in a range of habitats including cleared land, rainforest, pine forest, sedgeland and urban areas (Fielding, 1976; Fielding, 1979; Duckworth, 2001; Newman, 2002). Densities of this species in dry sclerophyll forest were found to be greater in young regrowth then mature forest and lowest in older regrowth (Taylor *et al.*, 1997).

Hollow requirements: Dusky robins breed occasionally in tree hollows but more often in low bushes, upturned trees, behind flaking bark, on tree stumps, branch forks and banks (Dove, 1910; Wilson, 1984; Newman, 2002). Green found 38 dusky robin nests (Australian National Wildlife Collection). For those where a description was provided, most were located in trees or bushes but two were in embankments and one was in a man made structure. Only three were specified to be hollows, two of which were in dead stumps (Australian National Wildlife Collection). Nests are usually located less than three metres above the ground (Dove, 1910; Newman, 2002).

Status and threats: This species is largely considered to be stable in numbers although some people have expressed their concern about a possible decline (Appendix 1). The main threats to this species are considered to be forestry, clearing for agriculture, competition for nest sites and predation (Appendix 1). However, reports suggest that this species is not afraid of people (Dove, 1910) and it still occurs in regrowth areas although at reduced densities (Green, 1982a; Wilson, 1984).

Flame robin (*Petroica phoenicea*)

Distribution: Flame robins are found across the state, although the majority of records are from the Midlands and the southeast (Thomas 1979). They are found in a range of habitats and breed in a narrow altitudinal range (Newman, 2002). This species requires a patchy environment and was found at lower densities in the uniformly thick undergrowth of older regrowth dry sclerophyll forest than in either young regrowth or mature forest (Taylor *et al.*, 1997).

Hollow requirements: As well as using hollows, flame robins nest in cavities in creek banks, in quarries, disused mines, among roots of upturned trees, cavities behind bark, wood piles, on the ground and in open sheds (Fletcher, 1924; Sharland, 1958; Napier, 1967; Newman, 2002). Tree hollows used by flame robins were found to regularly face east or north by Fletcher (1924).

Status and threats: A comparison of the two national Atlas surveys indicates a significant decline in flame robin populations with a decrease in reporting rates across the entire range of the species (Barrett *et al.*, 2003). There are also anecdotal reports of a decrease in their

presence around Hobart (see Newman, 2002). Land clearing, forestry and competition for nesting sites are perceived to be the major threats to the flame robins (Appendix 1). A decrease in abundance after clearfelling has been found in some studies (Coulson and Coulsen, 1981) although they have been found in cleared and regenerating areas (Green, 1982a). Another proposed threat is climate change and the corresponding change in rainfall patterns (Newman, 2002).

Forty-spotted pardalote (*Pardalotus quadragintus*)

Distribution: This species is a leaf-gleaning insectivore found only in Tasmania (Brereton *et al.*, 1997; Dorr, 1999). It is strongly associated with *Eucalyptus viminalis* dry forest which is an important food source (Woinarski and Rounsevell, 1983). This species is usually found in coastal areas below 100 m in altitude on fertile soils (Brown, 1986; Brereton *et al.*, 1997). They prefer undisturbed habitat near creeklines (Brown, 1989b). All known localities except that on Flinders Island are in southeast Tasmania, with the most important populations being on Maria Island and Bruny Island (Brereton *et al.*, 1997).

Hollow requirements: The degree of hollow dependency of the forty-spotted pardalote may vary between locations. In an examination of opportunistically located breeding sites, Brown (1986) found that most breeding occurred in north or east facing hollows in limbs of live *E. viminalis*. In contrast, Woinarski and Bulman (1985) found that on North Bruny Island they were found more often in alternative sites to tree hollows. Such sites include stumps, fallen trees or limbs and occasionally holes in the ground (Sharland, 1983; Woinarski and Rounsevell, 1983; Woinarski and Bulman, 1985; Brown, 1986; Brown, 1989b). Individuals often nest in close proximity (Sharland, 1983), with nests being as close as five metres and even occurring in the same tree (Woinarski and Bulman, 1985). The height of nests above ground vary from 0-38 m (average 9.2 m) (Woinarski and Bulman, 1985; Brown, 1986). Home range size is on average 0.7 ha (0.3 – 1.6) (Woinarski and Bulman, 1985). They are found at densities of 4.2 birds/ha in prime habitat and 0.6 birds/ha in secondary habitat on Maria Island (Brown, 1989b).

Status and threats: The forty-spotted pardalote is classified as endangered locally and nationally (Commonwealth *Environmental Protection and Biodiversity Conservation Act* 1999, Schedule 3 Tasmanian *Threatened Species Protection Act 1995*). While some reports indicate a decline in range since European settlement (Brown, 1986), recent modelling indicates that they might always have been a rare species with a restricted distribution (Brereton *et al.*, 1997; Bryant, 1997). About 60% of the populations occur in reserves (Brereton *et al.*, 1997), although on Bruny Island approximately 8% of the populations occur in reserves (Brown, 1986). The total population size is estimated to be between 3000 to 4600 individuals and currently stable (Threatened Species Section, 2006). Land clearing is probably the biggest threat to this species and recovery programs have included planting *E.*

viminalis. Wildfire has also been proposed as a threatening agent as it damages nesting sites in mature trees (Statham, 1984; Brereton *et al.*, 1997). They are also potentially threatened by competition to nesting sites as they have been observed defending their nests against potential competitors such as tree martins and green rosellas (Woinarski and Rounsevell, 1983; Woinarski and Bulman, 1985).

Striated pardalote (*Pardalotus striatus*)

Distribution: In Tasmania, records are concentrated in the southeast, but have been found across the state (Thomas, 1979). They are possibly common in grazing areas (Sharland, 1983). They prefer *E. viminalis* woodland with overmature trees but the association is not as strong as for the forty-spotted pardalote (Woinarski and Rounsevell, 1983).

Hollow requirements: Striated pardalotes generally use tree hollows for nesting but they can also use holes in the ground, caves and human structures (Appendix 1; Sharland, 1958; Sharland, 1983; Green, 1995). One anecdotal report was received saying that the preference for nesting site may have changed from tree hollows to nesting in holes in banks in northern Tasmania (see Appendix 1). Striated pardalotes sometimes nest in close proximity (Sharland, 1983) and nests can be clumped in distribution, being only two metres apart or in the same tree (Woinarski and Bulman, 1985). Green (1995) reported 30 pairs on the bank of the Tamar river in an area 30 m long and 3 m high with burrows about 40 cm deep. In the northeast of the state, nests are largely found in northeast facing trunk or branch hollows in live *E. amygdalina* and *E. obliqua* (Haseler and Taylor, 1993). They appear to prefer a relatively horizontal entrance located 0 to 28 (average 7.8) metres above the ground (Woinarski and Bulman, 1985). Hollow entrances are usually small (2.4-6 cm) and deep enough for nests to be set back from the entrance (Haseler and Taylor, 1993). They have been found at densities of 0.73 to 1.1 breeding adults per hectare (Woinarski and Bulman, 1985).

Status and threats: The striated pardalote is believed to be stable at reasonable population sizes (Appendix 1). It is found at lower densities in harvested forest than mature forest (Taylor *et al.*, 1997). Striated pardalotes are believed to be sensitive to logging (Appendix 1) although they have been located in regenerating forest (Green, 1977). Competition for nesting sites is also considered to be a problem, conflicts having been observed with forty-spotted pardalotes (Woinarski and Rounsevell, 1983).

Dusky woodswallow (*Artamus cyanopteris*)

Distribution: Dusky woodswallows migrate to Tasmania from the mainland to breed in open forest (Sharland, 1958). Most records are from the southeast of the state, with numbers

decreasing across the north coast. Only a few records have been found on the central west coast (Thomas, 1979).

Hollow requirements: Dusky woodswallows can use hollows for breeding but they often use stumps and bushes (Appendix 1; Sharland, 1958; Coulson and Coulsen, 1981).

Status and threats: This species is thought to be stable at low to moderate numbers although there is some concern that it may be in decline (Appendix 1). The Atlas data suggested that a decrease has occurred in eastern Tasmania and that an increase has occurred in the west (Barrett *et al.*, 2003). The biggest threats to this species are believed to be forestry, clearing for agriculture, predation and competition for nesting sites (Appendix 1). However the effect of forestry is uncertain with some studies finding a decrease in numbers after forestry activities (Wilson, 1981) and others finding them at greater density in regrowth than older forest (Taylor *et al.*, 1997).

Common starling (*Sturnus vulgaris*)

Distribution: Records of the common starling have been found across the state, although with few records in the southwest and some areas of the mid and northwest (Fielding, 1976; Thomas, 1979). They prefer areas near human settlement (Thomas, 1979; Harris, 1980).

Hollow requirements: Starlings use tree hollows for nesting but can also nest under roofs and in buildings (Appendix 1; Sharland, 1958). Of the 17 nests found by Green, 16 were in man made structures and one was in a dead stump (Australian National Wildlife Collection).

Status and threats: This species was introduced some time in the late 1800s (Green, 1995). Although it is believed by many to be increasing in numbers (Appendix 1), the Atlas data largely indicated no change in density between 1980 and 2000 (Barrett *et al.*, 2003). Being an introduced species, threatening processes are not considered. This aggressive species is one of the main competitors for native hollow-nesting birds (Appendix 1).

Sugar glider (*Petaurus breviceps*)

Distribution: Sugar gliders are found in most areas of Tasmania (Munks *et al.*, 2004a) but seem to be absent from the far southwest (Holdsworth, 2006). They typically prefer forest with 'old growth' elements (Statham, 1987; Slater, 1987a).

Hollow requirements: A study in the southern forests in Tasmania found that up to 20 sugar gliders may nest in hollow trunks and limbs of old growth trees (Slater, 1987a). However they have been observed nesting in other locations such as in the roofs of buildings (C. Spencer pers. comm.). They generally live in groups of four or five (but up to 20) in eucalypt trees (Statham, 1987) and their distribution has been described as 'patchy' within a forest stand (Slater, 1987a).

Status and threats: The sugar glider is thought to have been introduced to Tasmania in the 1800s (Gunn, 1851). Populations are currently considered stable. The potential threat of logging on sugar glider populations is uncertain. Although no sugar gliders were found in 52 year old regrowth in southern Tasmania (Slater, 1987a), they have been found in selectively logged and burnt forest at high altitude in the central highlands (Slater, 1987b). Introduced predators such as the feral cat can reduce sugar glider populations (Munks *et al.*, 2004a).

Eastern pygmy possum (*Cercartetus nanus nanus*)

Distribution: In Tasmania the eastern pygmy possum is found scattered across the state (Rounsevell *et al.*, 1991) in a wide range of habitats from rainforest through sclerophyll forest to heathland (Duncan and Taylor, 2001; Munks *et al.*, 2004a). They are believed to prefer mature habitat (Slater, 1987a) although they have been found in 20 year old clearfall regenerating forest (Duncan, 1995; Duncan and Taylor, 2001).

Hollow requirements: Eastern pygmy possums use tree hollows but can also utilize other spaces such as hollow logs, tree stumps and shredded bark in the forks of trees (Duncan, 1995; Duncan and Taylor, 2001). In southern Tasmania the majority of nests are under the bark of large trees (Slater, 1987a). At least three nests may be used by the same individual (Duncan and Taylor, 2001). The distance between nest sites for an individual range from 25-133 m (Duncan and Taylor, 2001).

Status and threats: Little is known about the current status of the eastern pygmy possum. Although there is no information indicating that they are declining, they may be at risk of decline in certain areas due to intensification of forest management (Munks *et al.*, 2004a).

Little pygmy possum (*Cercartetus lepidus*)

Distribution: Thought for many years to be confined to Tasmania, the little pygmy possum has now been collected from Victoria and South Australia (Green, 1979; Duncan, 1995). In Tasmania their distribution is widespread but with more concentrated records in the east (Rounsevell *et al.*, 1991). They are found in a range of habitats from heath to tall wet forest (Duncan and Taylor, 2001). They are thought not to occur in rainforest although anecdotal reports have been received (Wall, 1985). They are said to prefer more mature habitat (Slater, 1987a), although they have been found in both wet and dry regenerating forest (Duncan, 1995).

Hollow requirements: While little pygmy possums do use tree hollows, they also use decayed logs, hollow stumps, overturned turf, abandoned birds nests, fissures in trees, wall cavities, tree branches and clothes (Green, 1979; Duncan, 1995; Duncan and Taylor, 2001). Nesting places are usually close to the ground and often temporary in nature (Green, 1979). In the southern forests of Tasmania the majority of nests were found in or under thick bark at

the base of old *E. obliqua* trees (Slater, 1987a). The only nest site observed by forest contractors (Slater, 1987a) that was not in an old-growth tree was an 80 year old tree with exceptionally thick bark. Established nests have usually been found to have one or two adults (Green, 1979; Slater, 1987a). At least three nests may be used by one individual (Duncan, 1995; Duncan and Taylor, 2001). The observed distance between nest sites varies from 25-88 m (Duncan and Taylor, 2001).

Status and threats: The little pygmy possum is believed to be the rarer of the two pygmy possums found in Tasmania (Slater, 1987a). It has been proposed that they once occurred as a continuous population from the southeast to the north coast (Green, 1979). After European settlement, populations are thought to have declined due to habitat clearance and fragmentation (Green, 1979; Duncan, 1995; Hocking and Driessen, 1996). It is possible this species will experience a reduction in population size in the future but there is considered to be no risk of extinction (Munks *et al.*, 2004a). They are potentially preyed on by cats, quolls, owls and snakes (Green, 1979).

Bats

Distribution

All eight bat species in Tasmania are widespread and none are rare (Taylor *et al.*, 1987). Different bat species exhibit different foraging strategies, related to their body structure, which is likely to result in different distributions. Slow manoeuvrable flight is well suited for use in areas with obstacles such as dense vegetation, while faster less manoeuvrable flight is better suited to more open habitats such as above the canopy (O'Neill and Taylor, 1986). While species adapted to closed forest are able to utilize open environments, species adapted to foraging in open habitats are unable to utilize closed habitats, limited by the mechanical and sensory ability to deal with clutter. However, in Tasmania, O'Neill (1984) found that bat community diversity was not related to the structural diversity of the habitat, instead suggesting that the distribution and abundance of species may be dictated by the availability of roosts. In a study comparing bat density in five habitats in eastern Tasmania, coastal mallee had the greatest trapping success (seven species), followed by regrowth forest (six species), dry forest (five species), wet sclerophyll forest (three species) and grassy forest (two species: O'Neill, 1984). Diversity of bats has also been found to be high in blackwood swamp and low in rainforest (Taylor and O'Neill, 1985). Species diversity and bat density is thought to be lower in the west and southwest and higher in the east of the state (Taylor *et al.*, 1987; Taylor and Comfort, 1993). To date, only two species (lesser long-eared bat and King River vespadelus) have been recorded around Melaleuca (southwest Tasmania) while in eastern Tasmania eight species are likely to occur sympatrically (Taylor *et al.*, 1987). More trapping in the southwest may find other species, or the lower diversity could be a

result of smaller insect populations, greater rainfall or lower temperatures (Taylor *et al.*, 1987).

Hollow requirements

Bats use tree hollows for diurnal roosts and maternity colonies, but have been found roosting in logs on the ground, behind bark, in fence posts, buildings and in clothes (Green, 1966a; Statham, 1984; Taylor *et al.*, 1987; Taylor and Savva, 1988). There is only one record of bats using caves in Tasmania (Savva and Taylor, 1986). Although often found roosting under bark in forests, Australian forest bats are more reliant on tree hollows for maternity roosts (Churchill, S, 1998). Trees selected for roosting by bats tend to be mature, overmature or dead eucalypts (Taylor and Savva, 1988). They have been found to use a large height range of trees and prefer a diurnal resting place where they fit snugly between two surfaces. Entrance openings vary from 1.5 cm to a metre and they have been found to change roosts frequently (Taylor and Savva, 1988). The distance between roosts on successive nights ranged from 80 m to 1.4 km (Taylor and Savva, 1988). They both nest alone or in colonies of up to 26 individuals (Taylor and Savva, 1988).

Status and threats

All bat species in Tasmania have a wide distribution and none are rare (Taylor *et al.*, 1987). Their populations are believed to be stable but it is recommended that the endemic subspecies (greater long-eared bat) should be monitored because it is apparently less widespread than other species (R. Brereton pers. comm.). Little is known about the requirements and threatening processes for bats in Tasmania and more research is required. However the result of a national survey indicated that the priorities for conserving bats in Australia are currently: the effects of habitat fragmentation, effects of logging and the effects of mining (Lunney, 1989). Due to the limited amount of mining occurring in Tasmania, it is unlikely to have a great impact, but concerns have been raised about collisions with turbines on windfarms as evidenced by the "Bats and windfarms" brochure published by the Australasian Bat Society Inc. (2005). There are records of some individuals being killed by wind turbines in Tasmania, especially in autumn (R. Brereton, pers. comm.).

Bats can use a large range of habitat types and are sometimes thought to be less sensitive to fragmentation and deforestation than other species due to their ability to travel large distances (Tidemann and Flavel, 1987; Law, 1996). The ability of a particular species to survive in fragmented landscapes is likely to be influenced by the roosting ecology of that species and the availability of suitable roosting and foraging sites (Lumsden *et al.*, 1995; Lumsden *et al.*, 2002). Studies differ in the observed impact of forest fragment size on the species richness of bats, but linear strips and scattered trees can be used by some species on

mainland Australia (Lumsden *et al.*, 1995; Law, 1996; Lumsden *et al.*, 2002). The lack of a strong preference for particular vegetation types or structure in Tasmanian bats possibly makes them adaptable (at least in the short term) to landscape changes (Taylor *et al.*, 1987). Bats have been found foraging in regrowth forest due to a high abundance of insects (Taylor and O'Neill, 1985). Some studies have found a greater abundance of certain species in regrowth forest than other forest types (Taylor and O'Neill, 1985), while others have found greater bat activity in mature than regrowth forest (Rhodes, 1996). A study in Tasmania examined 23 roost sites. All were located in mature forest despite many of the bats being originally caught in regrowth forest (Taylor and Savva, 1988). This indicates that forestry practices are likely to have a detrimental effect on the availability of roosting sites for Tasmanian bats, as has been suggested for mainland Australia (Law, 1996).

Lesser long-eared bat (*Nyctophilus geoffroyi pacificus*)

Lesser long-eared bats are widespread and relatively abundant throughout the state (Green, 1966b; Rounsevell *et al.*, 1991). They are a relatively slow, highly manoeuvrable species capable of using dense vegetation (O'Neill and Taylor, 1986) and have been found in regrowth forest (Taylor and O'Neill, 1985). Their presence has been related to the number of dead stems in a forest (O'Neill, 1984). This species is thought to use buildings and houses for roosting more than other species, but they are still not frequent visitors (Green, 1966b). They are least abundant in regrowth and dry sclerophyll forest and most abundant in rainforest (Taylor and O'Neill, 1985) and mature forest (Rhodes, 1996).

Greater long-eared bat (*Nyctophilus timoriensis*)

This species was formerly believed to be *Nyctophilus gouldii* but is now recognized as *Nyctophilus timoriensis,* Tasmanian subspecies *sherrini* (Taylor *et al.*, 1987). It is probably endemic to Tasmania as a sub or true species (Taylor *et al.*, 1987). Its distribution is largely located in the eastern half of Tasmania (Bryant, 2002). This species is relatively slow and highly manoeuvrable, capable of foraging in dense vegetation (O'Neill and Taylor, 1986). Its presence is thought to be higher in areas with a high shrub cover and low eucalypt density (O'Neill, 1984). They are not commonly trapped, but have been found at low densities in a range of forested areas (O'Neill, 1984; O'Neill and Taylor, 1986). They are believed to be impacted by logging (Rhodes, 1996).

Gould's wattled bat (*Chalinolobus gouldii*)

Gould's wattled bats are largely located in north and central Tasmania (Bryant, 2002). They are not commonly trapped but have been found in blackwood swamp and wet sclerophyll forest at low densities (O'Neill and Taylor, 1986). They are thought to be the swiftest and

least manoeuvrable species in Tasmania (O'Neill and Taylor, 1986). In Tasmania they are occasionally killed by wind turbines (C. Hull, pers. comm.). They are initially affected by forestry practices but activity increases with forest age (Rhodes, 1996).

Chocolate wattled bat (*Chalinolobus morio*)

This relatively common species is thought to be widespread throughout the state but at greater densities in the north and east (Rhodes, 1996; Bryant, 2002). It is found in a range of forest types (Taylor and O'Neill, 1985; O'Neill and Taylor, 1986; Rounsevell *et al.*, 1991; Rhodes, 1996). It prefers deeper eucalypt canopies and smaller tree sizes (O'Neill, 1984). It has shown lower activity levels in regrowth forest (Rhodes, 1996).

Little forest eptesicus (*Vespadelus vulturnus* - formerly *Eptesicus*)

This is one of the most common species in Tasmania. Based on morphology, the little forest eptesicus is believed to use relatively closed habitats (O'Neill, 1984). They are most abundant in coastal mallee and blackwood swamp and are rare in dry sclerophyll, forestry regrowth and wet sclerophyll (O'Neill, 1984; O'Neill and Taylor, 1986). They occur in highest numbers at low altitudes (Rhodes, 1996). They are associated with *E. amygdalina*, greater shrub cover, patchy tree distribution, number of dead stems and lowland situations (O'Neill, 1984; Taylor *et al.*, 1987). They use tree hollows and buildings as roosts (Green and Rainbird, 1984).

King river vespadelus (*Vespadelus regulus* - formerly *Eptesicus*)

Probably the most common species in Tasmania (O'Neill, 1984; Taylor *et al.*, 1987), the King river vespadelus is believed to be widespread throughout the state (Rounsevell *et al.*, 1991). They prefer dense eucalypt cover (but not *E. pulchella*) and thick understorey, being rarely found in open habitats with patchy cover (O'Neill, 1984). They can be found in high numbers in regrowth forest (Taylor and O'Neill, 1985).

Large forest vespadelus (*Vespadelus darlingtoni* - formerly *Eptesicus sagittula*)

A common species in Tasmania, the large forest vespadelus is probably widespread throughout the state, particularly in the north and east (O'Neill, 1984; Rounsevell *et al.*, 1991; Bryant, 2002). They have average agility and are expected to prefer open areas (O'Neill and Taylor, 1986). Studies have found them to be most abundant in dry sclerophyll and regrowth sites (Taylor and O'Neill, 1985). Large forest vespadelus are trapped most often in dry forest, unlike the other two species in this genus which prefer mallee habitat (O'Neill, 1984). They have been associated with dense understorey and numbers of dead

stems (O'Neill, 1984). The effect of logging on this species is unclear and it has been found in both mature and regrowth forest (Rhodes, 1996).

Tasmanian pipistrelle (*Falsistrellus tasmaniensis*)

This species has largely been found in the eastern half of Tasmania (Bryant, 2002). Despite being one of the less commonly found species in Tasmania, Tasmanian pipistrelles can be found foraging and roosting in urban and city environments (O'Neill, 1984; Taylor *et al.*, 1987). Their presence is associated with greater shrub and *E. amygdalina* density and lower understorey patchiness (O'Neill, 1984). Their activity is potentially favoured by forestry practices as they are found in cleared and regrowth areas (Rhodes, 1996).

Medium hollow users

Fourteen species were identified as being able to use medium-sized hollows in Tasmania (Table 2.3). This included twelve bird species and two marsupials. Locality records for medium hollow users are prevalent across the state, with a similar species diversity distribution as was seen for small hollow users. That is, fewer species are found in the west of the state (Figure 2.2). Details on the specific habitats used by each of the species are provided in Table 2.3. One study on brushtail possums in an urban area found that up to 17 hollows may be used by one individual (Table 2.3). Mainland studies for several other species indicated between 0.004 and over 71 hollows per hectare may be used. No information was found on the spacing of nest trees required in Tasmania for any of the medium hollow using species although up to four ringtail possums may be found in the same nest. Mainland studies of these species indicated a range in tree spacing from as little as several metres to over a kilometre (Table 2.3). None of the medium hollow using species in Tasmania are threatened. The green rosella is endemic to Tasmania and the southern boobook, the eastern rosella and the two possums are subspecies endemic to Tasmania (Table 2.3).

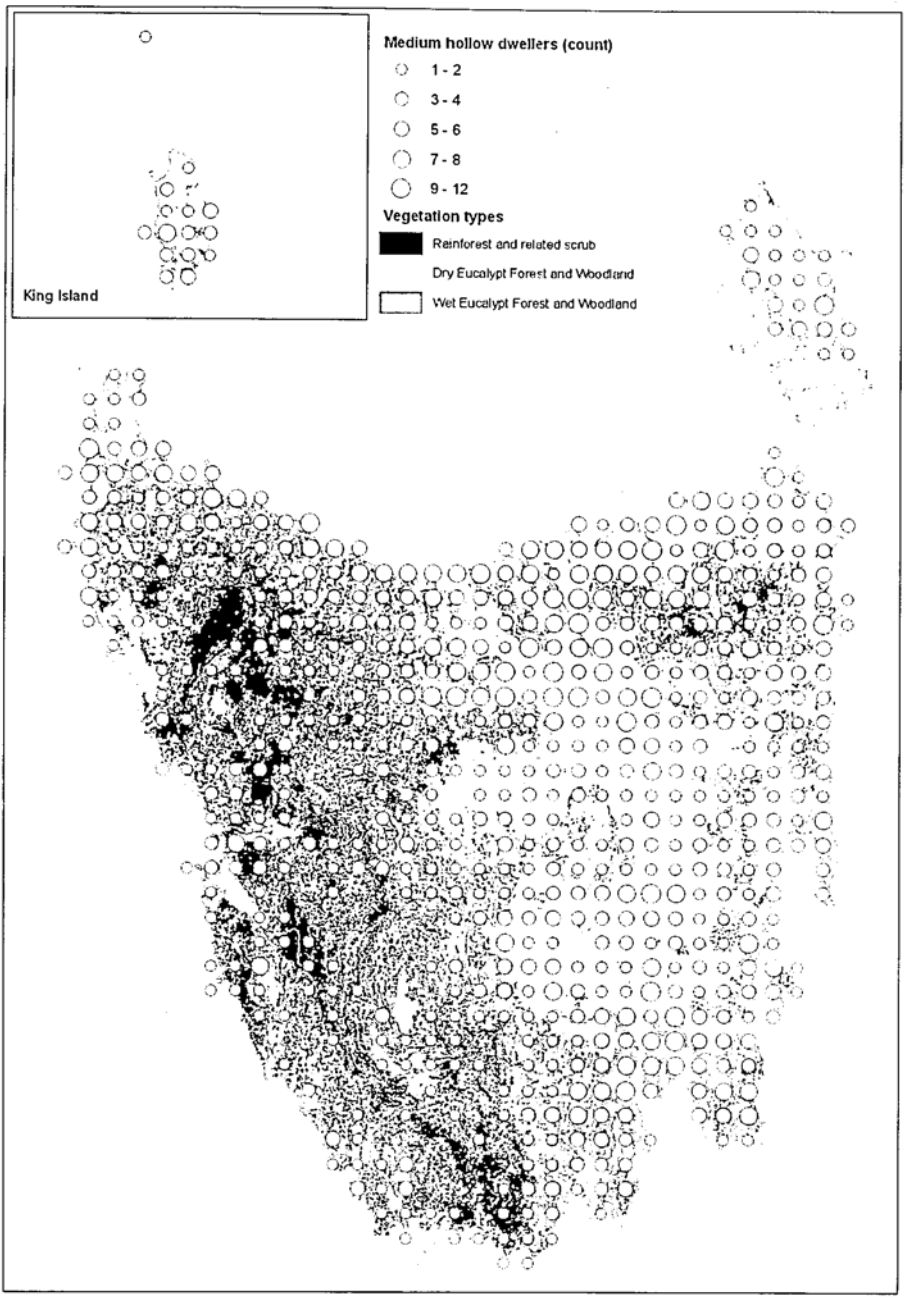


Figure 2.2 Map indicating the number of species able to use medium hollows that are known to exist in 10 x 10 km grid squares around Tasmania. Medium hollows have a minimum entrance width of 6-10 cm. Indication is also given of the type of forest found in the different areas of the state. Note: No records for the little corella were obtained from the Atlas data although they are known to occur in the central Tasmania.

Table 2.3 Conservation status, hollow use, habitat, population density, density and spacing of nest trees for animals that use medium hollows (6-10 cm entrance width) in Tasmania. Blank cells indicate no information was available.

Species	Status	Hollow use	Habitat type	Animal density (animals/ ha)	Number nest trees used per ha	Calculated spacing of nest trees (m)	References
Southern boobook owl	t	B S	W, D, Wd, S, C, A, SA, U, (R)	0.05-0.12 (T) 0.05 territories/ ha (M)		1000 (M)	Sharland, 1958; Recher *et al.*, 1971; Bosworth *et al.*, 1976; Thomas, 1979; Loyn, 1980; Blakers *et al.*, 1984; Bell *et al.*, 1997; Hollands pers. comm. in Higgins, 1999
Pacific black duck	-	R S, (G)	Fr, Td	11-38 nesting attempts in 2.5 ha (M)		Often close (M)	Sharland, 1958; Thomas, 1979; P.J. Fullagar and C.C. Davey pers. comm. in Marchant and Higgins, 1990
Australian shelduck	-	F S, (G)	Fr, (Td)			> 10 (M)	Thomas, 1979; Marchant and Higgins, 1990
Chestnut teal	-	B S, (G)	Fr, Td, C			In nest boxes can be close (M)	Thomas, 1979; Marchant and Higgins, 1990
Australian wood duck	-	F S, (G)	Fr, A				Foster, 2001
Grey teal	-	B S, (G)	Fr, Td, U	Up to 10 pairs in 2.5 ha of ponds (M) 0.06-0.22 (M)		Can nest in same tree (M)	Sharland, 1958; Frith, 1967; Thomas, 1979; P.J. Fullagar and C.C. Davey pers. comm. in Marchant and Higgins 1990; Marchant and Higgins, 1990
Galah	-	B G, (S)	Wd, A, U	0.07-0.11 (M)	3.2 (M)	10-80 (M) Defend hollow if others come within 3 m (M)	Saunders *et al.*, 1982; Lenz, 1990; Rowley, 1990; Green, 1995
Long-billed corella	-	B G, (S)	A			Up to five nests in one tree (M)	Saunders *et al.*, 1982; I.D. Temby pers. comm. in Higgins, 1999; Coupland, 2000
Little corella	-	B G, (S)	A, Wd		0.47 (M)		Saunders *et al.*, 1982; Coupland, 2000

Table 2.3 continued

Species	Status	Hollow use	Habitat type	Animal density (animals/ ha)	Number nest trees used per ha	Calculated spacing of nest trees (m)	References
Laughing kookaburra	-	B S	W, P, A, SW, D, Re, U	0.02 – 1.31(T) 0.03-0.23 territories/ha (M)	0.009 (T)	100–1500 (M)	Recher *et al.*, 1971; Thomas, 1979; Loyn, 1980; Taylor and Haseler, 1993; Green, 1995; Taylor and Haseler, 1995; Taylor *et al.*, 1997; S. Legge pers. comm. in Higgins, 1999; Duckworth, 2001
Eastern rosella	t	B S, (G)	SW, D, A, U, Wd	1.6-5.5 (M)		Av. 45 (M)	Thomas, 1979; Green, 1983; Green, 1995; Penck *et al.*, 1995
Green rosella	T	B S, (G)	CH, SW, D, W, MF, R, A, P, Wd, ML, U, (SA, CF, M, Re)	0.2-2.47 (T) 0.32 breeding pairs /ha (T)	0.004-0.04 (T) 0.5 nests/ha (T)		Recher *et al.*, 1971; Bosworth *et al.*, 1976; Brown, 1979; Thomas, 1979; Green, 1983; Wilson, 1984; Taylor and Haseler, 1993; Taylor *et al.*, 1997; Higgins, 1999; Hingston, 2000; Duckworth, 2001
Common brushtail possum	t	F C	R, W, D, A, U, Re	0.14-7.6 (T)	0.09-17 per individual unspecified area (T) 0.2-11 (M)	30-224 (M) More than 5 individuals can live in the same tree (M?)	Green, 1977; Johnson, 1977; Hocking, 1981; Statham and Statham, 1997; Smith and Lees, 1998; Gibbons and Lindenmayer, 2002; Munks *et al.*, 2004a
Common ringtail possum	t	F C	R, W, D, M, A, Re	1.5 – 2.47 (T)	3 (T) 0.4-71 (M)	Up to 4 individuals in the same nest (T) 12-158 (M)	Green, 1973; Green, 1977; Hocking and Guiler, 1983; Munks, 1990; Smith and Lees, 1998; Gibbons and Lindenmayer, 2002; Munks *et al.*, 2004a

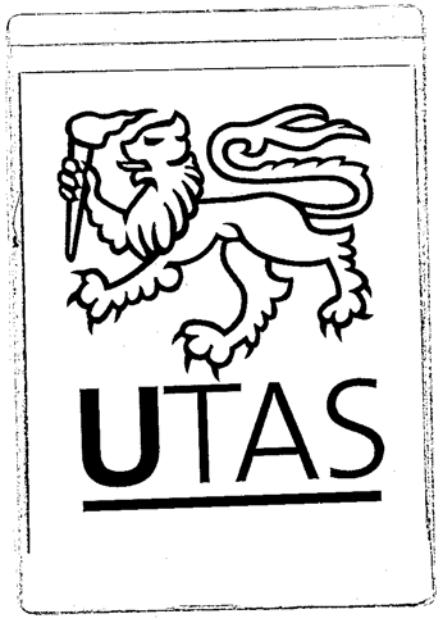
STATUS: M = migratory, T = Tasmanian endemic, t = Tasmanian subspecies is endemic (although sources differ in their decision on this), E and e = endangered under Commonwealth *Environmental Protection and Biodiversity Conservation Act 1999* (EPBC) and Tasmanian *Threatened Species Protection Act 1995* (TSPA), respectively. V and v = vulnerable under EPBC and TSPA, respectively. R and r = rare under EPBC and TSPA, respectively.

HOLLOW USE: *Degree of dependency*: BS = requires hollows for breeding and shelter (i.e. roosting or denning), B = breeding, requires hollows for breeding, F = facultative, uses hollows when available but can use other breeding and/or nesting sites, R = rarely, rarely uses hollows. (Note, even for the breeding-shelter and breeding categories, there have often been observations of nesting in non-hollows, but if the vast majority of instances are in hollows they are considered to be eligible for these categories). *Sociality of nesting*: G = gregarious, many pairs nest in close proximity, each requiring one hollow, S = single, each pair requires only one hollow which may be isolated from other members of the species, C = communal, hollow shared by more than one individual but each individual uses multiple hollows, M = multiple, multiple hollows are used by each individual but nesting is usually solitary. Brackets indicate this is done only occasionally.

HABITAT TYPE: R = rainforest, W = wet forest, D = dry forest, MF = mixed forest, Wd = woodland, CF = coniferous forest, ML = mallee, CH = coastal heath, C = coastal, S = sedgeland, SW = savannah woodland, M = moorland, Al = alpine, SA = subalpine, A = agricultural land, P = plantation, Re = regrowth, U = urban, Fr = freshwater, Td = tidal. Brackets indicate low densities.

CALCULATED SPACING OF NEST TREES: These figures represent the distances apart used nests have been found. For species that use multiple nests it may refer to distances between nests used by an individual. For species that only require one hollow per breeding pair, it will be the distance between nests of different individuals.

(M) indicates results are from mainland Australia, (T) indicates results are from Tasmania.



Southern boobook (*Ninox novaesseelandiae leucopis*)

Distribution: The southern boobook occurs throughout Tasmania at low to moderate altitude (Bell *et al.*, 1997). Although more abundant in the drier eastern half of the state, they can be found in a wide range of habitats (Bell *et al.*, 1997), being less common in alpine regions, extensive moorland, scrub and rainforest (Bell *et al.*, 1997). This species can use modified landscapes as long as sufficient nesting sites are available (Bell *et al.*, 1997). So although they are more likely to be found in older forest (Wilson, 1981), they have been seen in cleared and regenerating forest (Mollison, 1974).

Hollow requirements: Southern boobooks usually use tree hollows for nesting, but have also been recorded nesting in nesting boxes and other man-made structures (Bell *et al.*, 1997). They roost in tree hollows as well as dense foliage, rocky crevices, caves and man-made structures (Bell *et al.*, 1997). Bell *et al.* (1997) examined the eleven nesting records in Tasmania and all were in live or dead eucalypts. Of thirteen roosting sites, most were on the external surfaces of plants or in man made structures. Green examined four nests, which had a range in depth of 60-457 cm (Australian National Wildlife Collection). Home range sizes vary from 4 to 40 ha (Bell *et al.*, 1997).

Status and threats: The southern boobook is considered to be stable at low numbers (Appendix 1) although there is a chance they are declining (Bryant, 2002; Barrett *et al.*, 2003). Little is known about the population size of this species, although an estimate of 85,093 breeding individuals has been made based on home range size (Bell *et al.*, 1997). Southern boobooks are found in a number of reserves across the state (Bell *et al.*, 1997). The biggest threats to this species are considered to be forestry, the clearing of land for agriculture, firewood collection and competition for nest sites with starlings and kookaburras (Appendix 1; Bell *et al.*, 1997; Bryant, 2002). Collision with vehicles is also likely to be an issue although it is not clear how large a concern it is (Bell *et al.*, 1997).

Pacific black duck (*Anas superciliosa*)

Distribution: Pacific black ducks are found across the state, including the southwest (Fielding, 1976; Thomas, 1979) and are associated with lakes, rivers, lagoons and estuaries (Sharland, 1958).

Hollow requirements: Pacific black ducks usually nest on the ground but sometimes in tree hollows near water (Sharland, 1958; Green, 1995).

Status and threats: This species is believed to be stable at relatively high numbers (Appendix 1). There is some concern about the impact of legal hunting on this species (Appendix 1), but despite being the most heavily hunted species in the state, there is no sign of population decline (Game Management Services Unit, 2005).

Australian shelduck (*Tadorna tadornoides*)

Distribution: Australian shelducks are found in fresh water, some tidal water areas and pastures throughout the state but mostly in the eastern half including the highland lakes (Frith, 1967; Thomas, 1979).

Hollow requirements: This species uses tree hollows for breeding, although they can also use holes in the ground (Sharland, 1958) and rock crevices and disused rabbit burrows on islands and in treeless areas (C. Spencer pers. comm.).

Status and threats: The population size of this species appears to be remaining stable at reasonable numbers (Game Management Services Unit, 2005). There is concern about the impact of legal hunting as well as competition for nesting sites, forestry and land clearing (Appendix 1).

Chestnut teal (*Anas castanea*)

Distribution: Chestnut teal are found across the state including the highland lakes and the west (Frith, 1967; Thomas, 1979). On the mainland they generally breed in coastal areas (Norman and Brown, 1988) and commonly occur near rivers, estuaries, marshes and lakes (Sharland, 1958; Frith, 1967).

Hollow requirements: Chestnut teal sometimes nest on the ground, but more commonly nest in hollow trees in or near water (Sharland, 1958).

Status and threats: This species shows erratic population trends (Game Management Services Unit, 2005). They are available for hunting and in 2004 comprised 21% of the harvested ducks (Game Management Services Unit, 2005).

Australian wood duck (*Chenonetta jubata*)

Distribution: A survey by the Game Management Services Unit (2005) looked at 485 dams in all areas of Tasmania except the west coast and World Heritage area. They found an average of 1.3 Australian wood ducks per dam in the northwest, 2.5 birds per dam in the northeast and 3.3 birds per dam in the southeast. However, these figures are likely to vary upon repeated surveys and are not an indication of the range in bird numbers found per dam as Australian wood ducks generally occur in flocks. They have also been sighted in the southwest of Tasmania (Foster, 2001).

Hollow requirements: This species nests in holes in trees or on the ground among reeds (Sharland, 1958). In the southwest of Tasmania they roost near narrow, vegetation-lined, flowing streams (Foster, 2001).

Status and threats: Australian wood ducks were previously rare in Tasmania but they have increased in population size and distribution and they are now considered common (Green, 1995; Game Management Services Unit, 2005). A survey in 2004 (Game Management

Services Unit, 2005) counted a total of 1145 Australian wood ducks at 485 dams. The Australian wood duck was included in 2004 as a game species for a trial period of three years. First surveys suggest that hunting has not affected population sizes (Game Management Services Unit, 2005).

Grey teal (*Anas gracilis*)

Distribution: Grey teal live near freshwater and tidal areas (Thomas, 1979). They are distributed across the state, including the southwest, with greatest densities occurring around the Tasman peninsula (Fielding, 1976; Thomas, 1979).
Hollow requirements: Grey teal use tree hollows for nesting and generally roost on the ground (Australian National Wildlife Collection; Sharland, 1958).
Status and threats: This species is believed to be stable at low to moderate numbers (Appendix 1) although the observed densities are variable (Game Management Services Unit, 2005). This species is available for hunting (Game Management Services Unit, 2005).

Galah (*Eolophus roseicapilla*)

Distribution: There are a number of resident groups of galahs, largely along the north coast of Tasmania but with some further south and east (Brown and Holdsworth, 1992). They have been found in modified landscapes (Thomas, 1972).
Hollow requirements: It is thought that galahs use hollows mainly for breeding (Appendix 1) as they do in Western Australia (Rowley, 1990).
Status and threats: Galahs are thought to have been introduced to Tasmania, although it is possible that individuals can migrate from the mainland (Brown and Holdsworth, 1992). They are believed to currently be at low numbers (less than 500 individuals) but probably increasing (Appendix 1; Brown and Holdsworth, 1992; Barrett *et al.*, 2003). While a loss of nesting hollows could be an issue for this species, there is no great concern for its future in Tasmania (Appendix 1).

Long-billed corella (*Cacatua tenuirostris*)

Distribution: Long-billed corellas have been reported in small numbers scattered throughout the northwest, Midlands area and around Hobart (Brown and Holdsworth, 1992).
Hollow requirements: Nesting of long-billed corellas in Tasmania has not been confirmed but is believed to occur (Brown and Holdsworth, 1992). They are likely to be dependent on hollows for breeding and possibly also use them for roosting (Appendix 1).
Status and threats: The presence of this species in Tasmania is believed to be the result of aviary escapes with reported sightings occurring from the mid 1980s onwards (Brown and Holdsworth, 1992; Coupland, 2000). In 1992 there were approximately 100 known long-

billed corellas in the wild but they are probably increasing (Rowley, 1990; Brown and Holdsworth, 1992; Green, 1995). As they are considered to be an introduced pest, threatening processes are not considered.

Little corella (*Cacatua sanguinea*)

Distribution: A questionnaire survey indicated that little corellas have increased their range from the Midlands into the northwest, northeast and southern regions of the state. The results show this is linked to an increase in cropping practices (Coupland, 2000).

Hollow requirements: They are known to breed in woodland in the Midlands of Tasmania (Brown and Holdsworth, 1992). They are likely to be dependent on hollows for nesting but may also use them for roosting (Appendix 1).

Status and threats: This species is believed to be an aviary escapee. The first wild report was in 1982 and they are believed to be increasing in numbers (Appendix 1; Brown and Holdsworth, 1992). In 2000 there was estimated to be 600 individuals (Coupland, 2000). This species is considered to be an introduced pest and consequently no threatening processes are considered.

Laughing kookaburra (*Dacelo novaeguineae*)

Distribution: Laughing kookaburras were introduced in the north of the state and are now widespread, although with a limited distribution in the west (Thomas, 1979). They generally inhabit open woodland or open sclerophyll forest, but are found in other areas (Thomas, 1979; Green, 1995). There is some evidence suggesting that they are found in mature forest more often than regrowth (Taylor *et al.*, 1997).

Hollow requirements: Laughing kookaburras use tree hollows for nesting but not roosting, although they can also use tree stumps and cavities in buildings (Appendix 1; Sharland, 1958; Lloyd, 2002). Haseler and Taylor (1993) found three kookaburra nests, all in hollows with enlarged chambers facing northeast. Minimum entrance diameters were between 8.5 and 10 cm and hollow depth was 30 to 62 cm. Internal dimensions were roughly 20 to 30 cm in diameter. A branch 40 cm thick is believed to be the minimum size that can be used by this species (Haseler and Taylor, 1993).

Status and threats: Laughing kookaburras were first introduced in the early 1900s (Green, 1995) and have been increasing in numbers since then. Being an introduced species, there is no need to mention threatening processes, although possibly thousands of individuals have been killed in Tasmania by people hoping to conserve native fauna (Green *et al.*, 1988). It should be noted however that the laughing kookaburra provides a threat to other species. Being largely dependent on tree hollows for nesting, this aggressive species competes with native species for nesting sites (Lloyd, 2002).

Eastern rosella (*Platycercus eximius diemenensis*)

Distribution: Eastern rosellas are found in the eastern half of the state, with the most records from the southeast (Thomas, 1979; Bryant, 2002). They do not like to venture far from trees (Green, 1983).

Hollow requirements: They use tree hollows for breeding but not roosting (Appendix 1). Green examined four nests and found the hollows used were 30 to 76 cm deep (Australian National Wildlife Collection).

Status and threats: The distribution and population size of the eastern rosella is believed to have declined since European settlement (Green, 1983; Green, 1995). They were previously abundant in the Midlands but due to a loss of habitat, competition for hollows and being killed by farmers they are now no longer found in areas where they were once common (Green, 1983). The current status of this species is thought to be stable at low numbers or declining (Appendix 1). Land clearing and competition for nests are likely to be the main threats to this species, with the introduced common starling as the main competitor (Appendix 1; Green, 1983; Bryant, 2002). However, road kill, window collision and poaching may also be of concern (Appendix 1; M. Holdsworth pers. comm.).

Green rosella (*Platycercus caledonicus*)

Distribution: The green rosella is a Tasmanian endemic that is found throughout the state (Brown, 1979; Thomas, 1979). They are found in most habitats with trees, including pine forest and paddocks (Fielding, 1976; Wilson, 1984; Duckworth, 2001). They favour mature forest with a well developed understorey, although they can be found feeding but not nesting in regenerating forest (Wilson, 1984; Statham, 1987). In thinned regrowth with scattered old trees, 13.7 birds were found per kilometre transect, while in unlogged regrowth only 5.5 birds were found per kilometre (Taylor and Haseler, 1995).

Hollow requirements: Green rosellas are reliant on tree hollows for breeding but not roosting (Appendix 1; Sharland, 1958; Statham, 1984). They breed only in larger trees in mature forest (tree diameter 0.4 to 2 m: Wilson, 1984). They can use hollows of a range of sizes, entrance width varying from 4-22 cm (Wilson, 1984; Haseler and Taylor, 1993). They have been found using relatively deep (40 to 91 cm) branch and trunk hollows (Australian National Wildlife Collection; Haseler and Taylor, 1993). The minimum size for the nest chamber used is considered to be 50 by 15 cm (Wilson, 1984). Green rosellas are thought to use hollows located about 6 to 40 m above the ground (Wilson, 1984; Statham, 1987). There is some evidence to suggest they prefer northward facing hollows (Wilson, 1984; Haseler and Taylor, 1993).

Status and threats: The King Island subspecies *Platycercus caledonicus brownii*, is listed as vulnerable under Schedule 4 of the Tasmanian *Threatened Species Protection Act* 1995 due to it's limited distribution (50 km^2) and small population size (500 mature individuals), although competition for nesting sites is also a potential issue (Legge *et al.*, 2004). Although there is some belief that there has been a gradual decline in green rosella numbers on mainland Tasmania since 1940 (Green pers. comm. from Higgins, 1999), the green rosella is generally believed to be stable at high numbers (Appendix 1). The Atlas data indicates a potential increase in numbers although they were more likely to be detected in the second survey (Barrett *et al.*, 2003). Issues such as forestry, clearing for agriculture, window collision, road kill and competition with other birds should be noted (Appendix 1). Some reports indicate a positive effect of clearing (Brown, 1979), but this is most likely a result of their ability to feed in regenerating forest while they need older forest for nesting (Wilson, 1984). Taylor and Haseler (1995) found the relative rates of use between regenerating and mature forest varied. Starlings are probably the main species competing for nesting hollows, but the green rosella has a wider distribution than the common starling which may help its population stability (Green, 1983).

Common brushtail possum (*Trichosurus vulpecula vulpecula*)

Distribution: Prior to the 1940s, the brushtail possum was probably confined to mountainous and forested areas (Green, 1973). They are now relatively common in most places, although they are partly restricted by the absence of trees (Guiler, 1953). Brushtail possums are rarely found in sedgeland and rainforest, with a limited distribution in the southwest of the state (Hocking and Guiler, 1983; Hocking, 1990; Munks *et al.*, 2004a). There is some evidence to suggest that brushtail possums increase in density in areas disturbed by fire or logging (Hocking, 1981; Driessen *et al.*, 1990).

Hollow requirements: Brushtail possums use tree hollows for nesting sites where available but alternative sites are also used such as rabbit warrens, rock piles, fallen logs, dense undergrowth, artificial hollows or crevices in buildings (Statham, 1987; Munks *et al.*, 2004a). It should be noted that hollows used by brushtail possums will generally be of a larger entrance diameter than 10 cm although this size hollow can be used. A study of den use in an urban area showed that between one and three (but up to 17) day sites were regularly used, with 28% of den sites being in trees (Statham and Statham, 1997).

Status and threats: Numbers of brushtail possums have increased dramatically in the last few decades (Driessen and Hocking, 1992). This has been attributed to a decrease in hunting pressure (Driessen and Hocking, 1992). Currently, the brushtail possum is the most abundant and widespread arboreal marsupial species in Tasmania (Munks *et al.*, 2004a) and its status is considered secure. Approximately 60% of Tasmania is thought to provide suitable habitat for the brushtail possum (Hocking, 1990). Although brushtail possums are partly protected

under Schedule 4 of the *Wildlife Regulations* (1999) under the *National Parks Wildlife Act* (1970), they are considered a pest species in many areas of Tasmania and permits are available for hunting and baiting. To ensure the species is not being overexploited, spotlight surveys have been done annually since 1975 (Hocking, 1990; Callister, 1991; Driessen and Hocking, 1992). Large numbers are also killed each year by collisions with vehicles on roads (Mooney and Johnson, 1979).

Common ringtail possum (*Pseudocheirus peregrinus convoluter)*

Distribution: In Tasmania, the common ringtail possum is found in most areas but is absent from more open vegetation in the southwest of the state (low and open woodland, sedgeland and scrubland) (Hocking and Guiler, 1983; Slater, 1987a; Munks, 1990).

Hollow requirements: Common ringtail possums can use tree hollows or build nests (dreys) for shelter. The preference between the two is proposed as being related to the availability of suitable understorey and the availability of suitable hollows (S. Munks pers. comm.). The communal nests are usually built in thick scrub or low canopy areas of various tall shrub and tree species, including regrowth eucalypt forest (Statham, 1987).

Status and threats: The common ringtail possum was common in Tasmania up until 1940 when it underwent a rapid decline, possibly due to a disease and over-hunting (Munks *et al.*, 2004a). The ringtail possum is now a wholly protected species in Tasmania (Schedule 2 of the *Wildlife Regulations 1999* under the *National Parks and Wildlife Act, 1970*). Numbers have partially recovered (Green, 1973) and are thought to be stable. While Green (1982b) that there is likely to be a long term reduction in total population size due to habitat loss resulting from intensive forest management practices, there is no risk of extinction (Bekessy *et al.*, 2003; Munks *et al.*, 2004a). In some areas the ringtail possum may have to compete with the brushtail possum for refuge sites (Munks *et al.*, 2004a). Young ringtail possums are at risk of predation by feral cats, wedge-tailed eagles, brown falcons, grey goshawk and the masked owl (Green, 1982b; N. Mooney pers. comm. in Munks *et al.*, 2004a) .

Large hollow users

Three species are limited to using large hollows in Tasmania and cannot use hollows with an entrance diameter less than 10 cm. All three of these species are birds (Table 2.4). Clear patterns in the distribution of large hollow users are difficult to determine due to the low number of species. While observations are more concentrated in the north of the state, they clearly have a wide distribution (Figure 2.3). Details on the specific habitats used by each of the species are provided in Table 2.4. Only one study was found that considered the nesting requirements of a large-hollow using species in Tasmania. It was estimated that masked owls require a distance of approximately 1.5 km between nesting sites and use 0.03 trees per

hectare (Table 2.4). The masked owl is also endemic to Tasmania and listed as endangered under the Tasmanian *Threatened Species Protection Act* 1995 (Table 2.4). They are also being considered for listing under the Commonwealth *Environmental Protection and Biodiversity Conservation Act* 1999 (M. Holdsworth pers. comm.).

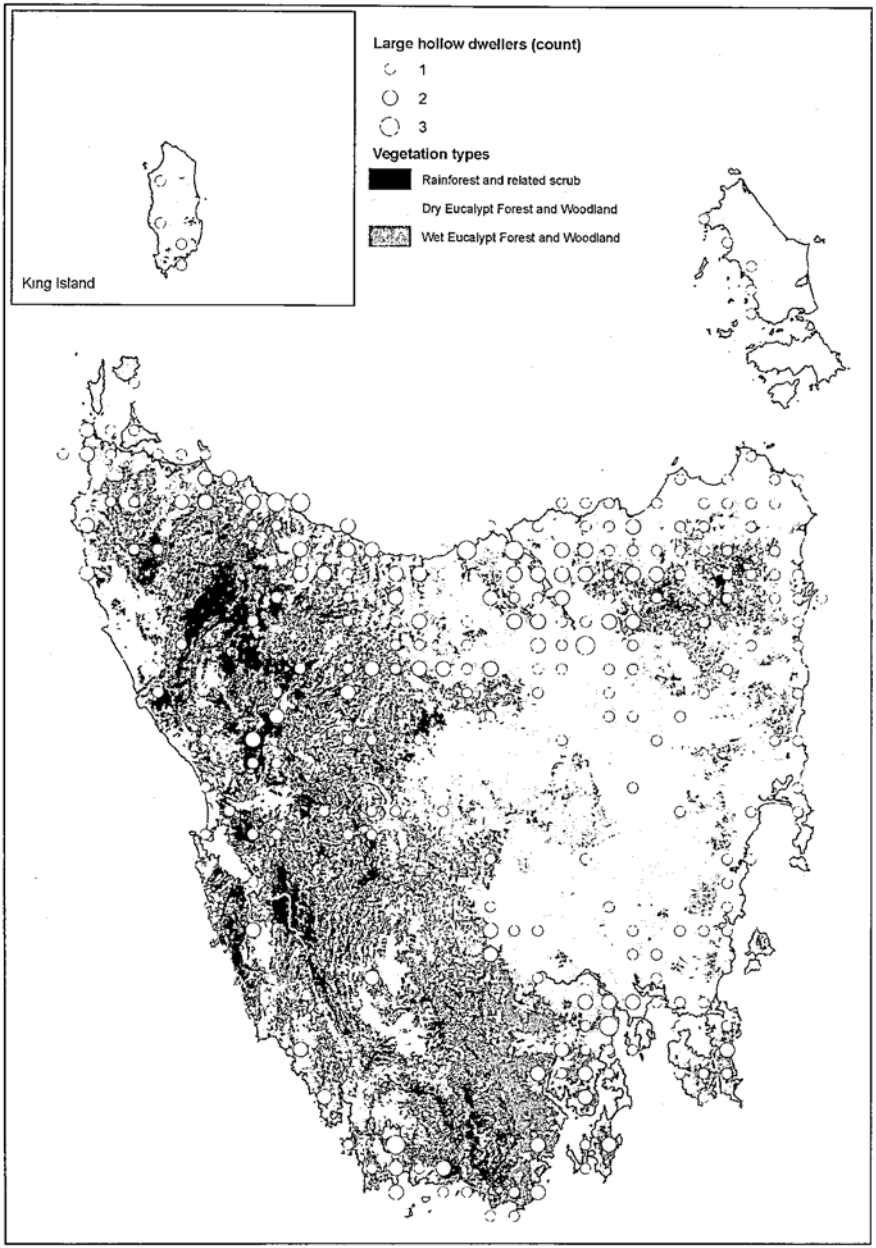


Figure 2.3 Map indicating the number of species able to use large hollows that are known to exist in 10 x 10 km grid squares around Tasmania. Large hollows have a minimum entrance width of more than 10 cm. Indication is also given of the type of forest found in the different areas of the state.

Table 2.4 Conservation status, hollow use, habitat, population density, density and spacing of nest trees for animals that use large hollows (>10 cm entrance width) in Tasmania. Blank cells indicate no information was available.

Species	Status	Hollow use	Habitat type	Animal density (animals/ha)	Number nest trees used per ha	Calculated spacing of nest trees (m)	References
Yellow-tailed black cockatoo	-	B S, (G)	R, C, W, SA, (SW, D, S, CF), P, CH, MF, A, U	0.05 (T) 0.13 territories /ha (M)	0.003 (T)	29 – 450 (M)	Sharland, 1958; Bosworth *et al.*, 1976; Green, 1977; Thomas, 1979; Loyn, 1980; Taylor and Haseler, 1993; Nelson and Morris, 1994; Green, 1995; Taylor and Haseler, 1995; Taylor *et al.*, 1997; Hingston, 2000; Wapstra and Doran, 2004
Sulphur-crested cockatoo	-	B S, (G)	W, R, S, A, MF, D, Wd			30 -500 (M) Can nest in same tree (M)	Sharland, 1958; Bosworth *et al.*, 1976; Green, 1977; Brown, 1979; Thomas, 1979; Frith and Frith, 1993; Green, 1995; Higgins, 1999; Hingston, 2000
Masked owl	te	B M	D, C, W, A, Wd			> 1500 (T) 150-8000 (M)	Thomas, 1979; Mooney, 1992; Debus, 1993; Kavanagh and Murray, 1996; Bell *et al.*, 1997; Mooney, 1997; Hollands pers. comm. in Higgins, 1999

STATUS: M = migratory, T = Tasmanian endemic, t = Tasmanian subspecies is endemic (although sources differ in their decision on this), E and e = endangered under Commonwealth *Environmental Protection and Biodiversity Conservation Act 1999* (EPBC) and Tasmanian *Threatened Species Protection Act 1995* (TSPA), respectively. V and v = vulnerable under EPBC and TSPA, respectively. R and r = rare under EPBC and TSPA, respectively.

HOLLOW USE: *Degree of dependency*: BS = requires hollows for breeding and shelter (i.e. roosting or denning), B = breeding, requires hollows for breeding, F = facultative, uses hollows when available but can use other breeding and/or nesting sites, R = rarely, rarely uses hollows. (Note, even for the breeding-shelter and breeding categories, there have often been observations of nesting in non-hollows, but if the vast majority of instances are in hollows they are considered to be eligible for these categories). *Sociality of nesting*: G = gregarious, many pairs nest in close proximity, each requiring one hollow, S = single, each pair requires only one hollow which may be isolated from other members of the species, C = communal, hollow may be used by more than one individual but each individual uses multiple hollows, M = multiple, multiple hollows are used by each individual but nesting is usually solitary. Brackets indicate this is done only occasionally.

HABITAT TYPE: R = rainforest, W = wet forest, D = dry forest, MF = mixed forest, Wd = woodland, CF = coniferous forest, ML = mallee, CH = coastal heath, C = coastal, S = sedgeland, SW = savannah woodland, M = moorland, Al = alpine, SA = subalpine, A = agricultural land, P = plantation, Re = regrowth, U = urban, Fr = freshwater, Td = tidal. Brackets indicate low densities.

CALCULATED SPACING OF NEST TREES: These figures represent the distances apart used nests have been found. For species that use multiple nests it may refer to distances between nests used by an individual. For species that only require one hollow per breeding pair, it is more likely to be the distance between nests of different individuals. (M) indicates results are from mainland Australia, (T) indicates results are from Tasmania.

Yellow-tailed black cockatoo (*Calyptorynchus funereus xanthanotus*)

Distribution: Yellow-tailed black cockatoos are thought to be widely distributed and are found in most habitats around Tasmania (Thomas, 1979), even being found at low numbers in pine plantations, forest remnants and in the west (Brown, 1979; Brown and Holdsworth, 1992; Duckworth, 2001). They breed only in mature forest (Wilson, 1981), possibly breeding more in wet forest than dry (Brereton pers. comm. in Fox and Brereton, 2003) and breeding birds are believed to be intolerant of disturbance (Brown and Holdsworth, 1992).

Hollow requirements: Yellow-tailed black cockatoos are reliant on large hollows for breeding (Appendix 1; Brown and Holdsworth, 1992). They often use dead trees or trees with large dead limbs (Brown and Holdsworth, 1992). Large hollows are required due to the large size of the bird, with reported holes being 20-30 cm wide and 65 cm deep (Haseler and Taylor, 1993; Wapstra and Doran, 2004).

Status and threats: The yellow-tailed black cockatoo is found at low numbers and some people are concerned they are declining (Appendix 1). This belief is disputed by others, but the large size and loud call of the species and their gregarious and nomadic nature means they are readily noticed (Brown and Holdsworth, 1992) which, in conjunction with their relatively long life span, may delay recognition of a decrease in numbers. Although there is no immediate concern, further research may be warranted. One report was received indicating that numbers are increasing in the north of the state, which was attributed to their having learnt to eat pine cones (Appendix 1). The only estimate of population size in Tasmania is around 100 breeding pairs in the northeast of the state (Fox and Brereton, 2003). The results of some modelling work predicted declines under all scenarios that involve anthropogenic disturbance, being driven by a reduction of breeding habitat (Fox and Brereton, 2003).

Sulphur-crested cockatoo (*Cacatua galerita galerita*)

Distribution: Sulphur-crested cockatoos are widespread and common. Their range is growing in Tasmania (Coupland, 2000), although they may have a limited distribution on the east coast, the northeast in particular (Brown, 1979; Thomas, 1979; Coupland, 2000). They are thought to prefer open wooded country (Statham, 1987), although they have been found in rainforest, buttongrass (Mollison, 1974; Brown, 1979), wet forest (Brown and Holdsworth, 1992), in suburban areas (Thomas, 1972) and orchards (Fielding, 1979).

Hollow requirements: Although sulphur-crested cockatoos are believed to be reliant on tree hollows for breeding, there are few breeding records for this species in Tasmania (Appendix 1; Brown and Holdsworth, 1992). They have been observed to occupy hollow limbs in trees, preferring the more mature and large trees with a hollow entrance size of 20 cm or more (Brown and Holdsworth, 1992). They often roost communally but show little indication of

being selective in terms of tree species or site, although patch size may be important (Coupland, 2000). The main factors influencing choice of nest site in Tasmania are suggested to be food availability and distance required to travel to feed (Coupland, 2000).

Status and threats: The sulphur-crested cockatoo is believed to be widespread and common and possibly increasing (Appendix 1; Brown and Holdsworth, 1992; Coupland, 2000). There has been some discussion if this species is introduced or not, but reports indicate it is native to Tasmania (Brown and Holdsworth, 1992). This species is believed to be stable at reasonable numbers and no threats are considered to be of immediate concern, although potential issues are a decrease in nesting sites and habitat and competition for nest sites (Appendix 1; Pattemore, 1980).

Masked owl (*Tyto novaehollandiae castanops*)

Distribution: The masked owl is generally found throughout the state in a range of habitats although there are fewer records from the west (Bell and Mooney, 2002). There is evidence that they occur at higher densities in areas of low elevation, low rainfall and high annual mean temperature (Bell and Mooney, 2002). They are frequently found close to cleared or non-forested areas in areas with a mosaic understorey with open and dense areas (Bell and Mooney, 2002). Rainforest is of low importance to this species (Mooney, 1992).

Hollow requirements: This species usually nests in tree hollows, although there is a record of them breeding in a building and in a cave (Sharland, 1958; Bell *et al.*, 1997; Mooney, 1997). They can roost in hollows but also use other locations such as cliffs, caves, vegetation, and occasionally man-made structures (Mooney, 1992; Bell *et al.*, 1997; Bell and Mooney, 2002). Nests are rarely less than 1.5 km apart (Mooney, 1992) and home range sizes are estimated to be around 400-1000 ha (Bell *et al.*, 1997). Hollows used by masked owls are on average 9.5 m (2.8-26 m) above the ground in trees 1.24 m (0.95-1.56 m) in diameter, often with badly burned bases on the trees (Mooney, 1997; Wapstra and Doran, 2004). Hollow entrance widths of 30 cm have been recorded (Green, 1982b).

Status and threats: This species is listed as endangered at the state but not the national level (Schedule 3 Tasmanian *Threatened Species Protection Act* 1995). Populations in Tasmania are thought to be stable at low numbers or declining (Appendix 1). There is no direct evidence of a decline although anecdotal reports have been received saying masked owls were more commonly seen in the 1940s and 1950s (Bell *et al.*, 1997; Mooney, 1997). The population size was estimated at 1300 breeding individuals in 1997 (Bell *et al.*, 1997) and 200-400 pairs in 2002 (Bell and Mooney, 2002). The biggest threats to the masked owl are the forest industry, clearing for agriculture, lack of available nest sites due to attrition of nest trees and competition with bees, brushtail possums and kookaburras and collision with motor vehicles and powerlines (Appendix 1; Bell *et al.*, 1997; Mooney, 1997). Masked owls are sensitive to direct disturbance at the nest site (Bell *et al.*, 1997), although they have been

known to use isolated trees in pastureland (Mooney, 1997). Only a small area of the preferred habitat for masked owls (6%) is within reserves, most being on private land (Bell and Mooney, 2002).

Discussion

Hollow-using species in Tasmania

This review found that around 42 species of arboreal marsupials, bats and bird species use tree hollows consistently in Tasmania. This number drops to 35 if the introduced bird species and sugar glider are not included. The number of hollow-using species in Tasmania is at the lower end of the range found on mainland Australia; 42 to 70 species (see Gibbons and Lindenmayer, 2002 for collation). In comparing the rates of hollow use for the different taxa in Tasmania with rates on mainland Australia, the results are variable. It has been estimated that about 91 terrestrial birds have been recorded breeding in Tasmania (census by Bureau of Flora and Fauna, cited by Taylor, 1991). This means that the proportion of birds that use hollows in Tasmania (31.9%) is greater than the national average (15% when seabirds and migratory birds are excluded: Gibbons and Lindenmayer, 2002). The available information indicated that 35% of the 37 species of mammal in Tasmania (34 natives and three introduced rats or mice) (Watts, 1993) use hollows, which is slightly higher than the national average of 31% (Gibbons and Lindenmayer, 2002). Although this review did not cover amphibians and reptiles, a previous review by Munks *et al.* (in press) states that one frog and two lizards occasionally use hollows. If these records are correct then the rate of hollow use for amphibians (9%) and reptiles (9.5%) in Tasmania is roughly similar to the national rates of 13% and 10%. Therefore, while the number of species found to use tree hollows in Tasmania is at the lower end of many regions of mainland Australia, the proportion of species within a taxa that use hollows is similar or higher than the national average.

Within more local areas the number and taxa of hollow-using vertebrate species found in Tasmania may be similar to other temperate areas of Australia. For example, in southwest Western Australia 42 vertebrate species use tree hollows; seven marsupials, nine bats, 21 birds and five reptiles (Abbott and Whitford, 2002). Although the number of marsupials and bats found in southwest Western Australia is similar to that found in Tasmania, more birds and fewer reptiles were estimated to use hollows in Tasmania compared with southwest Western Australia. However, estimates of hollow-using species in Western Australia did not include species that were thought to not be part of the original forest fauna. If Tasmania's introduced species and recent natural migrants are excluded, the number of hollow-using birds in Tasmania is reduced to 23 species. Another difference in the composition of hollow-using fauna between the areas is that more large hollow users are found in Western

Australia; nine species have a maximum adult body width of at least 100 mm compared to the three large hollow users in Tasmania. These results indicate that despite the geographical isolation of Tasmania, in general the number and variety of hollow-using species is similar to that found in southwest Western Australia.

Distribution

Locality records (Figures 2.1, 2.2 and 2.3) suggest that fewer hollow-using species are found in the west of Tasmania. Some species, such as the orange-bellied parrot, are largely confined to the west of the state. However, a greater number of species are known to largely occur in the east of the state. These include the forty-spotted pardalote, the striated pardalote, the swift parrot, the eastern rosella and the musk lorikeet. The bird atlas data used to construct the distribution maps was collected during standardised surveys (Barrett *et al.,* 2003) which will limit the amount of bias that may occur between the geographical areas of the state and lends credence to the maps provided. However, given the limitations of the survey effort used in the atlas data, birds which occur in small numbers may have been missed during the surveys. Only a small number of additional sightings were obtained from expert ornithologists. Furthermore, much of the mammal distribution data was obtained opportunistically. Fewer people live in the west of the state which results in reduced observation effort in this area. This means the distribution data for the mammal species considered will be extremely biased towards areas of higher population, which may suggest a difference in distribution where none exists. However, published literature appears to confirm the trend of greater species richness in eastern Tasmania compared to the west. A greater number of bat species have been found in the eastern part of the state compared to the west (Taylor *et al.,* 1987; Taylor and Comfort, 1993), sugar gliders appear to be absent in the southwest of the state (Holdsworth, 2006), brushtail possums were not found in a survey in the southwest and ringtail possums were not found in open formations in the area (open woodland, sedgeland and scrubland) (Hocking and Guiler, 1983). It therefore appears that despite the shortcomings available in the data, the pattern of species richness suggested in Figures 2.1, 2.3 and 2.4 may reflect a real effect of a greater number of hollow-using species in the east of the state compared to the west.

The reported higher species richness found in the eastern half of the state roughly corresponds with forest type, with the western third of the state having more rainforest, wet sclerophyll forest and heathland in comparison to the rest of the state which is predominately dry sclerophyll forest and non-forest areas. Forest type may influence the ease with which animals are observed and so may also influence the rate of incidental reports of animal occurrence. Therefore, again, it is uncertain if the figures presented in this chapter reflect a preference for particular forest types by hollow-using fauna in Tasmania or not. Examination

of Table 2.5 suggests that there is little difference in the number of species found between wet forest and dry forest, although there is possibly a decrease in species richness in mixed forest and rainforest. However, this summary is again based on limited information and does not indicate how frequently species are found in the respective forest types. It has previously been reported that the number of forest-using birds are known to decrease from dry forest, through wet forest to rainforest (Statham, 1984). This suggests that the observed change in the number of hollow-using species is likely to decrease also through these forest types, as was suggested by the figures presented in this chapter, but firm conclusions cannot be made due to the limited nature of the data available.

Hollow requirements

As a generalization, the size of hollow used by an animal is related to its body size (Gibbons, 1999). The information gathered in this review indicated that three of the possums, all eight bats and 14 birds (total 25 species) can potentially use small hollows (Table 2.1). Two possums and 12 birds (total 14 species) are medium hollow users and three birds are more than likely restricted to using large hollows (although it is likely that the bats are also reliant on large hollows for roosting colonies). Many of these species can also use hollows that are larger than the sizes indicated. Although there are many more small hollow users than large hollow users, many of the small hollow users can also use alternative nesting sites, such as holes in the ground, hollow logs, dense undergrowth, tree stumps etc. In comparison, all three bird species limited to using large hollows are reliant on hollows for breeding (Table 2.6a).

This dependency on tree hollows for breeding is found for the majority of the hollow-using bird species in Tasmania (Table 2.6b). This means that most bird species will require only one hollow per breeding pair during the breeding season. An exception to this is the Australian owlet nightjar, which is the only hollow-user in Tasmania that is dependent on hollows for both nesting and roosting. Some other species which breed in hollows, such as the masked owl, the southern boobook and the little corella, also use hollows for roosting but they can use alternate locations. In comparison, all of the arboreal marsupials use hollows as a diurnal refuge and multiple hollows are used by each individual throughout the year, but all species can use alternate locations (Table 2.6b). Like the arboreal marsupials, Tasmanian bats use tree hollows all year round but can also use alternate locations. However, bats are believed to be more reliant on tree hollows for breeding sites than roosting sites (R. Brereton pers. comm.; Table 2.6b).

Table 2.5 A summary of the number of small, medium and large hollow users that have been reported in different habitat types. [a]

Species group	Habitat type											
	Mallee	Woodland/ savannah woodland	Dry forest	Wet forest	Mixed forest	Rainforest	Subalpine	Sedgeland /moorland	Regrowth	Plantation	Agriculture	Urban
Small hollow-users	9	15	20	17	5	13	4	10	10	1	15	8
Medium hollow-users	1	6	6	5	1	4	2	3	4	2	10	7
Large hollow-users	0	3	3	3	2	2	1	2	0	1	3	1
Total	**10**	**24**	**29**	**25**	**8**	**19**	**7**	**15**	**14**	**4**	**28**	**16**

[a] This information is a summary of the habitat information provided in Tables 2.2, 2.3 and 2.4. It is a summary of the habitats in which hollow-using animals were sighted as determined by the literature examined. It is possible that species may be found in habitats not indicated in the literature. Furthermore, these figures do not reflect the relative densities of hollow-using species in these habitats or the distribution of breeding areas.

Table 2.6 A summary of the hollow dependency of hollow-using vertebrate fauna in Tasmania.

	Species group	Hollow dependency [a]				
		Breeding-shelter	Breeding	Facultative	Rare	Total
a)	Small hollow-users	1	15	6	3	**25**
	Medium hollow-users	0	9	4	1	**14**
	Large hollow-users	0	3	0	0	**3**
b)	Birds	1	19	5	4	**29**
	Arboreal marsupials	0	0	5	0	**5**
	Bats	0	8	0	0	**8**
	Total	**1**	**27**	**10**	**4**	**42**

[a] Breeding-shelter: requires hollows for breeding and for shelter (i.e. roosting, denning). Breeding: requires hollows only for breeding. Facultative: uses hollows when available but can use other breeding, roosting or denning sites. Rare: rarely uses hollows. (Note that some species in the breeding-shelter and breeding categories have also been observed using alternative sites, but the majority of observations are of tree hollow use).

Conservation considerations

The major processes identified as causing a decline of the hollow resource across Australia since European settlement are logging, clearance for agriculture, firewood collection, urban development, grazing and altered fire regimes (Gibbons and Lindenmayer, 2002). All these processes occur in Tasmania with habitat loss through agricultural and forestry practices and rural residential development being considered the greatest threat to habitat for hollow-using fauna. Road kill, wind farms and hunting are three other threatening processes considered in Tasmania. The incidence of animals killed by vehicles on the road (road-kill) is estimated to be higher in Tasmania than in any other state (A. Hobday pers. comm.), but the majority of road-kill species are non-threatened or non-hollow using fauna.

Production forestry operations are perceived to be the most serious threat to habitat for hollow-using fauna (Appendix 1). Forestry operations range from the selective harvest of dry eucalypt forest to the more intensive clearfelling of wet eucalypt forest. Conversion of native forest to plantations or agriculture is also currently part of production forestry in Tasmania. Conversion removes habitat permanently. In comparison, forest habitat can be maintained in the longer term by selective harvesting and clearfelling procedures, although the structure and age of the forests is changed. However, few hollows are likely to form given current rotation times used in most areas of Australia (e.g. see Ball *et al.*, 1999), there is more potential to ensure a continual supply of hollows in production forestry areas if some sort of selective harvesting or tree retention is undertaken. There is limited reservation of habitat within the areas of the state identified as potentially having the highest densities of hollow-using species (Figure 2.4). Approximately 60% of the forest in Tasmania was potentially available for logging in 2003, including more than 70% of the more productive tall forests (Hickey and Brown, 2003). This amount has reduced to 54.4% (DPIW, 2001; DPIW, 2006) since the Community Forest Agreement (Commonwealth of Australia and State of Tasmania, 2005). However, the decrease in the amount of forest available for wood production without a corresponding decrease in harvesting quotas leads to an intensification of harvesting in the remaining forest (Lindenmayer and Franklin, 2002, p. 93). This means that the management of tree hollows in timber production areas is becoming increasingly important.

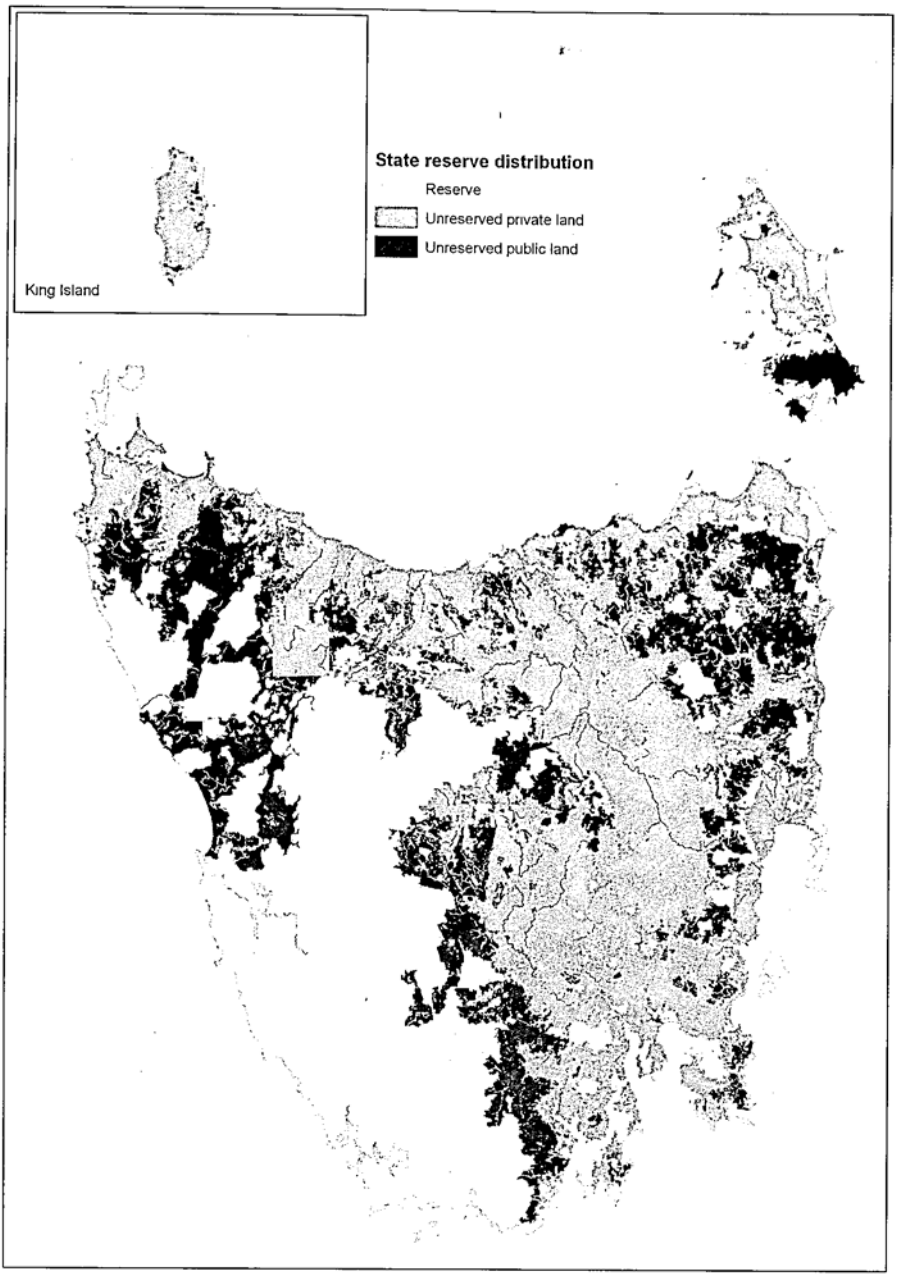


Figure 2.4 Map indicating the land tenure in Tasmania.

Future research directions

There is a lack of information available on the nest tree requirements for a number of hollow-using species in Tasmania, including the spacing of nest trees and the characteristics of the hollows used. Such information is required for assessing the adequacy of tree retention prescriptions, particularly in production forest areas. Making an assessment of the current management strategies is extremely important to ensure that a continual supply of tree hollows is available into the future for all fauna. However, an assessment of how to maintain hollow-bearing trees in areas that are not used for timber production (e.g. agricultural areas, urban areas) would also be extremely valuable. It also needs to be established the degree to which hollows are limiting populations of hollow-using species in Tasmania. This would involve a more thorough investigation of competition between species for the hollow resource and the current trends in exotic species such as bees and starlings. For those species that are not reliant on tree hollows, it would be useful to have more information on the consequences of using alternative nesting sites on fecundity and survival rates.

Conclusion

In Tasmania, 42 species use tree hollows to varying degrees. The five possums may use hollows for denning at any time, but all of these species can use other locations. All of the eight bat species can also use a variety of roosting sites although tree hollows are believed to be the preferred location for breeding. Of the 29 bird species, the Australian owlet nightjar (*Aegotheles cristatus*) was the only species which was reliant on tree hollows for both breeding and roosting. Nineteen species were reliant on tree hollows for breeding but not roosting, five species generally use tree hollows for breeding but can also use other locations and four species use tree hollows only occasionally. Four of these bird species are classified as endangered at the state and/or federal level (Commonwealth *Environmental Protection and Biodiversity Conservation Act* 1999, Tasmanian *Threatened Species Protection Act* 1995), all of which are reliant on tree hollows for breeding. Although the number of hollow-using species found in Tasmania is at the lower end of the range found in other areas of Australia, the proportion of species in a particular taxa that use hollows is similar or greater. Hollow-using fauna form a major component of Tasmania's vertebrate fauna.

There is some evidence to suggest there is a greater number of hollow-using species in the eastern half of the state, possibly associated with areas of dry forest areas rather than wet forest or rainforest areas. Dry forest areas are more heavily used by people for urbanization, forestry and agriculture. Consequently, the availability of hollows in these areas may be limited. Management of the hollow resource is particularly important in these areas, as well as in areas of the state where forests are poorly reserved (e.g. southeastern Tasmania).

There is a limited amount of data available on the specific nesting requirements of hollow-using fauna in Tasmania. Yet from the information that is available, it is apparent that the Tasmanian species have extremely diverse requirements. The number and spacing of nest trees required by a species varies according to the purpose for which hollows are used and, to varying degrees, with the size of the animals and the territoriality of the species. Most of the bird species only use hollows for breeding. Consequently, they will generally require only one hollow per breeding pair during the breeding season. In comparison, the arboreal marsupials use hollows as a diurnal refuge and multiple hollows are used by each individual throughout the year. The spacing of nesting sites used by a species varies from as little as one metre to as much as 1.5 km. The largest spacing of nests currently identified is required by the largest hollow-using animal, the masked owl. The smallest distance is required by one of the smallest species, the forty-spotted pardalote. The swift parrot is larger than the forty-spotted pardalote, but also uses hollows in close proximity due to the gregarious nesting habits of this species.

In terms of the types of hollows required by Tasmanian fauna, the majority of hollow-using species (59.5%) can use small hollows (≤5 cm in entrance diameter). Thirty-three point three per cent can use hollows that are medium in size or larger (6-10 cm) and 7.1% are limited to only the largest of hollows (>10 cm). However, many of these species, bats and brushtail possums in particular, will use hollows that are larger than their minimum requirements. Larger hollows take longer periods of time to develop. Therefore, despite the apparent low demand for large hollows in Tasmania, it is these hollows which need to be managed most carefully, particularly because all three large hollow users are dependent on hollows for breeding and one of these species, the masked owl, is threatened.

Chapter 3

Estimating the error of tree ageing methods in mature *Eucalyptus obliqua* forest, Tasmania

Abstract

Estimates of tree age are important for making management decisions on the tree hollow resource because hollows suitable for fauna occur with greater frequency in older trees. The methods used to age trees generally involve a combination of ring counting and extrapolating (when a section of the tree radius is missing) or the use of growth models (when no wood sample is available). This paper uses all three methods, including three different growth models, to understand the magnitude of the error associated with estimating tree ages. The error of extrapolation was related to the proportion of sample that was missing. Simple regressions between tree age and diameter provided the most accurate growth models. However, age estimates obtained from such models had unacceptably large errors when few trees were used or when variable growth rates occurred. Under these circumstances, a model based on tree diameter and site attributes should be used in preference to the average growth rates across sites. However, ring counting is the most accurate of the methods available and is recommended for use, with validation by cross-dating where possible. The error of ring counting was approximately 10% of the estimated age. Although the error margins can be large for the oldest trees (average ± 42.4 years for trees >350 years old), it was less than ± 15 years for most (72.8%) of the trees that were estimated to be between 100 and 300 years of age. These middle-aged trees are often the most useful to study when examining the rate of hollow production in eucalypts. Therefore, age estimates acquired in this way are generally accurate enough to be useful for making management decisions regarding the tree-hollow resource in production forests.

Introduction

Tree hollows provide important habitat for fauna and are produced through the action of fire, termites and fungi (Gibbons and Lindenmayer, 2002). Not only does an increase in time allow greater opportunity for these processes to act on a tree and produce hollows, but older trees are less effective at healing after wounding which means there are more and larger hollows in older trees (Mackowski, 1987; Whitford, 2002). Given the short time period between harvest rotations in production forest (60-120 years: Whiteley, 1999; Department of Natural Resources and Environment, 2002a; Department of Natural Resources and Environment, 2002b), in comparison to the time required for usable hollows to form (e.g.

150 years: Mackowski, 1987; 165 years: Wormington and Lamb, 1999; 130 years: Whitford and Williams, 2002), management of this resource in production forest areas requires an understanding of the average age at which trees in different areas produce hollows suitable for use by animals.

A number of studies in Australia have examined the rate at which hollows form in trees and the minimum age at which a tree can be used by fauna (Wormington and Lamb, 1999; Gibbons *et al.*, 2000b; Whitford, 2002; Whitford and Williams, 2002). A range of different techniques have been used to determine tree age, including ageing based on disturbance history (Bradshaw and Rayner, 1997b), tree ring counting to directly estimate the age of the tree (Banks, 1997) and using ring count or diameter increment data to produce a model predicting the age of a tree (Lloyd and Lau, 1986; Wormington and Lamb, 1999; Gibbons *et al.*, 2000b; Moloney *et al.*, 2002). However, all of these methods have errors. The magnitude of error involved in making age estimates has the potential to influence the adequacy of management prescriptions. For example, if large error margins exist but the average values are adopted by managers, hollows may be thought to occur in an area when they do not. This may result in harvesting regimes that do not allow sufficient time for suitable hollows to develop in retained trees. It is therefore important to both select an appropriate method for estimating tree age and assess the accuracy of the technique used so that the uncertainty can be accommodated in management prescriptions.

Ring counting is commonly used for ageing trees because it provides a good balance between cost and accuracy. This method relies on the assumption that there is annual variation in the density or size of the cells that are accumulated by the tree. The age of the tree is then estimated by counting the number of dark latewood bands. These darker bands are produced because there is a decrease in the growth rate of the tree due to seasonal conditions such as cold temperatures or moisture deficit. However, factors such as defoliation by grazers and fire can also influence the production of these bands (Mazanec, 1968; Mucha, 1979). Consequently, the production of latewood bands is not always strictly annual, with some rings being locally absent, or extra 'false' rings being produced. The rate of false ring production has been shown to vary with the dominance status of the tree, while locally absent rings occur most frequently on trees that are old, have poor crown development or where environmental stresses are high (Brookhouse, 1997).

Ageing trees by ring counting is especially problematic in eucalypts because the bands generally lack clear definition (Brookhouse, 1997) and the older trees often have a rotten or hollow middle. Ring clarity in eucalypts is expected to be greatest in areas where interseasonal climatic variation is greatest (Mucha, 1979). The problem of the missing rings

in the tree centre is generally solved by using the size of the rings found in the solid wood to estimate the number of rings on the missing sample (Rose, 1993; Woodgate *et al.*, 1994). The error in ring counting is usually assessed by cross-dating distinct rings or markings either between trees or with known disturbance events (Rayner, 1992; Brookhouse, 1997). However, only very rarely is reference made to the error associated with the extrapolation. Where such error is referred to, it is generally very vague, indicating that the accuracy of age estimates diminishes in older, rotten trees and no estimate of the error is provided (Rayner, 1992). Woodgate *et al.* (1994) was an exception, providing an upper and lower estimate of the number of rings to be added to a sample. In this case, the extrapolation was done by using the number of rings found in a complete cross section of an equivalent radial interval. The upper and lower estimates were obtained when multiple tree data was available. The error margins ranged between 13 and 17 years for trees estimated to be 137 to 237 years old.

Tree-ring counting is a useful method for estimating the age of a tree but is time-consuming and relies on the ability to obtain a suitable wood sample. Growth models are an alternative technique to estimate the age of a tree. For these models, known tree ages, growth rates or tree increment widths are used to make an estimate of tree age based on the diameter of the tree (Bradshaw and Rayner, 1997a; Gibbons *et al.*, 2000b). They generally work on the assumption that there is a constant growth rate in trees and are quick and easy to use and are especially useful when no wood sample is available. However, growth rates are known to vary with a number of factors, including tree age, dominance and species, environmental factors, tree density and site quality (Mucha, 1979; Rose, 1993; Wormington and Lamb, 1999). Some studies cater for this variability by using more complex models, such as having different relationships for different sized trees or forest types (Abbott and Loneragan, 1983). Studies which ignore this variability have attracted criticism by other authors who describe the resulting models as 'unsound and biased', particularly when estimating the age of the largest tree in the stand (Stoneman *et al.*, 1997). Caution in using these models is often advised (Abbott and Loneragan, 1983; Rose, 1993), but few studies indicate the amount of error actually produced. Some papers using ring-width data to produce growth models provide an indication of the error of these widths, but do not indicate the error of the age estimate itself (Abbott and Loneragan, 1983; Rose, 1993). Other studies have used cross-checking with available maps to test the stand age estimates made (Lindenmayer *et al.*, 1999), which, while not providing an estimate of the error, does provide more credibility to the ages used.

When considering the management of hollow-bearing trees in production forests, it is important to know the magnitude of error in age estimates. Gross over or under-estimations of tree age could have large consequences on the effectiveness of policies for managing the

habitat for hollow-dependent fauna. The aim of this paper is firstly to develop an appropriate growth model for estimating the age of *Eucalyptus obliqua* trees in Tasmania and secondly to investigate and outline a method for approximating the error of the age estimates when using either ring counting or a growth model.

Methods

Site selection and data collection

The study sites were forestry coupes containing mature wet or dry *Eucalyptus obliqua* forest. Sites were selected by availability, with 38 coupes examined between January 2004 and May 2005 (Figure 3.1). One tree was haphazardly selected at each site, with a further 11 or 12 trees selected according to the following criteria: the closest tree to the one previously examined that was at least 50 cm in diameter and at least 20 m from all other trees examined. Tree and site variables were measured (see Table 3.1 for details). The trees were then felled as part of normal forestry operations. A slice of wood was cut from the stump of 329 trees (between six and twelve trees per site). The best slice possible was taken given the condition of the stump. However, it was not always possible to take a complete sample (which contained the tree centre) or a consistent aspect for each tree due to physical limitations of the equipment and the condition of the stump. Many stumps were rotten or hollow in the middle.

All samples were planed and sanded as required for clear identification of tree rings. Tree rings were identified by the denser and darker coloured latewood bands. The number of tree rings was counted where they were most clear on one radius for each sample. Generally, only a wedge of wood was cut from each tree and the size of the sample prevented additional counts from being done. Terminology specific to this paper is used throughout the text and a summary of the more important terms is provided in Table 3.2.

Figure 3.1 Location of study sites in Tasmania, Australia

Table 3.1 Tree and site attributes measured.

Site and tree attributes	Description
Dbh	Tree diameter at breast height (1.3 m) over bark measured using a diameter tape (cm).
Tree shape	A classification of the shape of a tree, based on definitions by Smith and Lindenmayer (1988): (3) tree with full round crown; (4) mature tree with large branching; (5) mature tree with dead branches; (6) mature tree with dead top but big crown; (7) mature tree with dead top and small crown; (8) burnt tree with only epicormic growth.
Crown class	The relative dominance of each tree was categorised as: (1) dominant; (2) co-dominant; (3) sub-dominant or suppressed.
Tree height	The height of the tree was measured after felling (m).
Burn damage	The burn damage to the tree was assessed as: (1) no evidence of fire; (2) bark damage; (3) wood exposed and damaged by fire; (4) severe fire damage where the base of the tree forms a bridge.
Tree species	Tree species was determined.
Forest type	The forest type was determined using the RFA classification available from the Forest Botany Manual (Forest Practices Authority, 2005) and then reduced to the classes: Dry (DRY hOB); Damp (DRY shOB); Wet (WET all types).
Latitude	A GPS was used to determine the latitude of the sites in GDA. Values were confirmed using a map of the area.
Longitude	A GPS was used to determine longitude of the sites in GDA. Values were confirmed using a map of the area.
Aspect	A compass was used to determine the direction directly down-slope. The aspects were divided into four categories (N, S, E, W).
Altitude	A GPS was used to determine the altitude of the sites (m). Values were confirmed using a map of the area.
Topography	The average topography of the site was categorised as: (1) ridge; (2) upper slope; (3) mid-slope; (4) lower slope or gully.
Slope	A clinometer was used to determine the slope of the site in degrees.
Rock	Parent rock type of the substrate, was obtained from the plans developed for the harvesting of the coupe (Forest Practices Plans) and was classified as: (1) granite; (2) dolerite; (3) sediment.
Soil type	Soil type as obtained from Forest Practices Plans and was classified as: (1) loamy; (2) clayey; (3) sandy.
Soil pH	Four soil samples were collected from the top 10 cm. The samples were mixed and the pH was measured using a probe in a 1:5 soil: distilled water solution.
Soil nitrogen	Soil was air-dried, ground, sieved and measured for total nitrogen using the Kjeldahl method (Jackson, 1964).
Soil phosphorus	As for soil nitrogen, but measured for extractable phosphorus using the Bray and Kurtz method (Jackson, 1964).
Stand height	Categorised according to the information from the photographic interpretation (PI) of the vegetation age structure within each plot into: (0) unknown; (1) 20 m; (2) 30 m (3) 37 m; and (4) 50 m.

Table 3.1 continued

Site and tree attributes	Description
Stand age	Categorised according to the information from the photographic interpretation (PI) of the vegetation age structure within each plot (obtained from Forestry Tasmania's concise -PI type maps: Stone, 1998): (1) mature; (2) mature with regrowth; (3) regrowth with mature; (4) regrowth.
Average temperature	Annual mean temperature values obtained from ESOCLIM (McMahon *et al.*, 1995) using data on the latitude, longitude and altitude of the site.
Rain	Annual mean precipitation values obtained from ESOCLIM using data on the latitude, longitude and altitude of the site.
Radiation	Annual mean radiation values obtained from ESOCLIM using data on the latitude, longitude and altitude of the site.

Method for estimating tree age

For each wood sample, the number of rings counted, the length of the sample and an estimate of the distance missing on that radius to the centre of the tree was recorded. Often the middle of the tree could be seen when cutting the sample, but it was not possible to include it because of rot or the distance was greater than the bar of the chainsaw. In such cases there was high accuracy in determining the missing distance of the radius. When trees were hollow in the middle, the missing distance was estimated as best as possible based on the curve of the rings that were visible. Samples that contained the tree centre will be referred to as 'complete samples' and those with a section missing as 'incomplete samples'. The distance between every ten rings was measured from the inside out to determine ring width or 'increments' (Table 3.2). For each tree the relationship between the radius and the number of 'increments' was plotted and a regression line passing through the origin was fitted. The slope of this relationship will be referred to as the 'Increment Formula.' For incomplete samples, the Increment Formula was used to estimate the number of rings likely to have occurred in the missing section of wood. Relationships other than linear were explored for all fitted lines in this study but the linear relationship proved to be the best fit, producing plausible results in all cases, unlike the curved extrapolation. For further details see the methods section on extrapolating. Tree ages estimated using this technique are referred to as Ring Count And Extrapolation ages (RCAE ages) (Table 3.2).

There are a number of error sources when estimating the age of trees that are considered in the current paper (Figure 3.2). When a wood sample is available there is error from counting the number of tree rings on the wood sample and extrapolating the number of rings found on the missing centre of a tree. When there is no wood sample available and a growth model is used, the accuracy of the age estimate may depend on whether growth rates of other trees at the site have been examined or not.

Table 3.2 Definitions of terms used throughout the text.

Term	Definition	Sampling Details
RCAE Method	The method for estimating tree age, where ring counts are done and the age of the missing middle is estimated using linear extrapolation.	
RCAE age	The age of the tree as estimated using the RCAE Method.	329 samples were used to construct RCAE ages. Between 4 and 12 samples were examined for each of 38 sites.
Increments	The distance between every ten rings on a wood sample.	
Increment Formulas	The slope of the fitted line between the radius and the number of rings for each tree, used to extrapolate the age of the missing middle.	329 increment formulas were constructed. Between 4 and 40 increments were used to construct each formula.
Extrapolation Error Model	The model predicting the absolute percentage error in tree age based on the percentage of wood sample that was missing.	6 wood samples from dry forest that were not used in any other part of the current study were used to construct this model.
Site Formulas	The models predicting tree age, produced from the Bayesian regression analysis between the tree diameter and estimated age for all trees at a site.	38 site formulae were constructed. Between 4 and 12 trees were used to construct the formula for each site.
Overall Formula	The model predicting age of a tree produced from the Bayesian regression analysis between the tree diameter and estimated age for all trees combined.	329 samples were used to construct this model.
Predictive Model	A growth model predicting the age of a tree produced by the Bayesian regression model between the RCAE age of a tree and site and tree variables.	329 samples were used to construct this model.

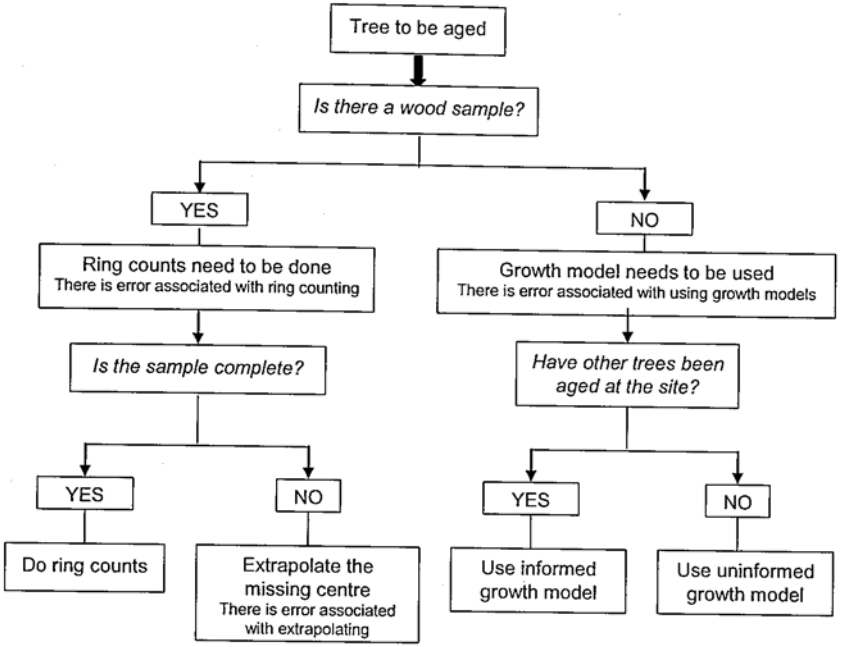


Figure 3.2 A schematic diagram illustrating how the age of a tree is determined and the associated error. The 'informed' growth model means that information is available on the growth rate of trees at the site and 'uninformed' means that no such information is available and the average growth rate across all sites is used.

Error in ring counts

In this study, tree rings were assumed to be annual, although the error made by this assumption is considered. Error in tree ring counts can occur as a result of false rings, missing rings, or observer error. Greater error is expected for workers unfamiliar with eucalypt growth rings (Mucha, 1979). To assess the accuracy of the ring counter in this paper, the rings of sixteen *E. obliqua* samples were counted by Koch and two other experienced counters. There was an average difference of 7% (4.5 s.d., max 17.8, min 1) between the number of rings counted by Koch and the first experienced ring counter and an average of 16% (16.2 s.d., max 55.6, min 1.2) with the second. However, the difference between the two experienced ring counters was an average of 16% error (maximum 60), suggesting that the error of the current author was not greater than is found among experienced counters. Another method for estimating the error in ring counts is to cross-date the estimated age with known disturbance events. This was not possible due to a lack of accurate information on disturbance events in Tasmania. Examination of data available indicated reasonable accuracy in ring counts, but results are not presented because the age estimates of disturbance events were broad or uncertain. As it was not possible to accurately cross-date the tree ages in this study, error rates for ring counting will be estimated as those

calculated by Brookhouse (1997). Brookhouse (1997) examined *E. obliqua* and *E. cypellocarpa* trees of known age in Central Gippsland, Australia and used cross-dated ring widths to examine error in ring counts. The results of Brookhouse (1997) were considered to be appropriate for use in the current study because the same tree species was examined, samples were taken from high altitude sites (680 – 700 m a.s.l.) where the climatic conditions are expected to be more similar to Tasmanian conditions, and the understorey species were similar to those found in the wet forest sites considered in the present study. Brookhouse (1997) found that ring counts overestimated the age of the tree by 3% in co-dominant trees, 7% in dominant and 8% for suppressed trees.

Error in extrapolating

To estimate the number of missing rings on an incomplete sample according to the distance of wood that was missing, a straight line was fitted to the growth increment data and this line was extrapolated back to the origin. Extrapolation using a quadratic function was also examined but the difference in R^2 value between the two fits was on average 0.0129 (s.d. 0.0189, max 0.124) and was not significantly related to any tree variable examined. However, the Increment Formulae were significantly associated with both tree age and shape ($p < 0.05$), suggesting that there is a deceleration in growth rates with increasing age and senescence. Some of the trees examined showed accelerating growth rates, some decelerating and others a combination of the two which made the uniform application of a decelerating function inappropriate. In the vast majority of cases a straight line was an adequate (and often more appropriate) fit for the data than a quadratic curve.

The data collected during this study could not be used to establish the error of extrapolating the missing length of wood because the complete samples collected were not from older trees. Consequently, data collected from another source in dry forest in eastern Tasmania was used (von Platen, unpublished). Six slabs were cut from large, complete *E. obliqua* tree stumps and treated as for the current study (see Table 3.3 for tree details). A high-resolution photograph was then taken and a computer program was used to identify growth rings, which were then verified either by Koch or von Platen. On obtaining the increment data, ten rings were removed at a time and the number of rings in the 'absent' area was estimated using linear extrapolation as outlined above. The percentage change in age estimate was examined in relation to the percentage of the sample that was removed. Linear regression analysis was done using SPSS (2005), where the dependent variable was the absolute percentage difference in age estimate and the independent variable was the percentage of sample that was removed. The resulting regression model, which will be referred to as the Extrapolation Error Model (Table 3.2), could then be used to predict the percentage error of the extrapolation for the other trees examined in this study, according to the proportion of the

wood sample that was missing. This percentage error was multiplied by the estimated age of the tree to obtain an estimate of error in years. This value was then added to the error found from ring counting in order to estimate the total error resulting from using the RCAE Method to age trees.

Table 3.3 Details of the trees examined for establishing error in linear extrapolation.

Tree number	Ring count	Sample radius (cm)	Tree diameter (cm)
1	220	62.6	140
2	270	57.4	125
3	230	55.9	165
4	221	64.3	130
5	155	46.0	120
6	146	32.6	100

Examining the fit of the RCAE Method

Having developed a method for estimating the age of a tree and the associated error when a wood sample is available, it then becomes important to assess the technique for problems or unexpected results. This was initially done by visually examining the relationship between error magnitude and tree age. Age estimates were then considered in light of expected results. Eucalypts are believed to be relatively short-lived, reaching only about 400-500 years in general (Rayner, 1992; Banks, 1997; Bradshaw and Rayner, 1997a; Brookhouse, 2006). If the general limit of 400 years is taken, there is a chance that trees estimated to be older than this in the current study may not be as old as is estimated. To examine possible sources of excessive error in these 'older' trees, photographs, notes on the ease of tree ring counting and the form of trees were examined as available.

Growth models and associated error

Modelling the influence of site and tree variables on growth rates

To identify the most important variables influencing the growth rate of individual trees, Bayesian linear modelling with uninformative priors was done in WinBUGS 1.4 (Spiegelhalter *et al.*, 2003). Two wood samples were not used to establish the growth models due to a lack of information on tree shape and size pre-felling. The RCAE age of the tree (square-root transformed) was the dependent variable. The independent variables considered are outlined in Table 3.1 and all continuous variables were standardised to reduce autocorrelation between successive samples (i.e. the mean was subtracted from the data, which was then divided by the standard deviation). To do the model selection, variables were entered in a forward stepwise manner. The initial 1,000 samples were discarded as a 'burn

in' and the following 10,000 samples were used to calculate the Deviance Information Criterion (DIC). DIC values can be interpreted in a similar way to AIC values (Burnham and Anderson, 2002) and were used for model selection. A difference in DIC value of less than two indicates a lack of difference in the models, while a difference of three or more indicates that the model with the smaller DIC value is increasingly superior (McCarthy, 2007). In addition, the variable 'site number' was added as a random factor to determine the degree to which variation occurred between sites, that could not be attributed to the site variables measured. For the final model (excluding site number), 100,000 samples were used to calculate the mean, standard deviation, 2.5^{th} and 97.5^{th} percentile of the coefficients. The percentiles represent a 95% Bayesian credible interval. The residuals (observed value minus predicted value) were examined to assess the fit of the model. The results of this model were then used to produce the 'Predictive Model', which predicts the age of a tree based on site and tree characteristics.

One of the variables expected to have an important influence on tree growth rates is forest type. Tree growth rates vary with site quality (Abbott and Loneragan, 1983), tree species and management history (Northern NSW Forestry Services, 2000), all variables which can vary between forest types. To determine if a difference in growth rate was observed between the broad forest types considered in the current study, ANOVA analysis was done between the slope of the Increment Formula and the three forest types.

Examining alternate growth models

The Predictive Model is one way of estimating the age of a tree based on diameter. However, it is common to use a regression style technique between tree age and diameter to produce a simple model for predicting tree age (see Bradshaw and Rayner, 1997a; Gibbons *et al.*, 2000b). Consequently, Bayesian regression analysis was done between tree age and diameter for all trees combined to produce the 'Overall Formula' and for the trees within a site to produce a series of 'Site Formulas' (Table 3.2). The three growth models (Predictive, Overall and Site models) were used to predict the age (and standard deviation of the estimate) for all the trees that were used to construct the models. The accuracy of the models was examined in three ways. Firstly the percentage of trees where the RCAE age fell within one standard deviation of the predicted age was determined. Then regression analysis was done between the RCAE age and the predicted age for each model using SPSS and the slope coefficient and the adjusted R^2 values were examined. Finally the error of the predictions was inspected by visually examining the relationship between the standard deviation and the predicted age for each tree.

Study limitations

There were a number of limitations to the accuracy of this work. The time to reach the height at which the wood slice was taken for aging was not taken into account. Alcorn *et al.* (2001) estimated that *E. obliqua* trees grow to 0.3 - 0.5 m by age one in wet forest in southwest Tasmania, which is less than the error margins for most trees examined. No account was taken of swelling in the base of the tree disproportionate to the size of the trunk (butt swell) because normal forestry procedures needed to be used. Trees exhibiting butt swell may have had some leverage on the relationship detected between age and diameter of trees. Large influence may also have been produced by samples taken on radii where there was a lot of compression or a lot of expansion. In terms of the Extrapolation Error Model, the trees used in its construction were only from dry forest in eastern Tasmania and so their relevance to growth in wet forest is uncertain. However, neither forest type nor latitude were found to significantly influence the growth rate of trees in this study, justifying the use of this model. Finally, the same data was used to produce the growth models as to test them in order to ensure a large enough sample size existed both to create the models and to determine the full range of accuracy that using these models produced. Ideally, a new data set should have been used to verify the accuracy of these models.

Results

Wood samples for age estimation were taken from 38 sites. A minimum of four wood samples were taken at each site, making a total of 329 samples, including 71 complete samples. The slope of the Increment Formula ranged in value from 0.09 to 1.05 between trees.

Error in extrapolating

The error in estimating the age of the tree increased with the percentage of the wood sample that was removed, but not in a consistent direction until approximately 70% of the sample was removed. When more than 70% of the sample was removed there appeared to be an increase in the rate of error for most trees, with ages being more consistently overestimated. The relationship between the absolute percentage change in age estimate and the percentage of sample that was missing was roughly linear (Figure 3.3). The model showed a reasonable fit, (average difference between predicted and actual error was 3.15), but the 95% confidence intervals showed a lack of precision in the results.

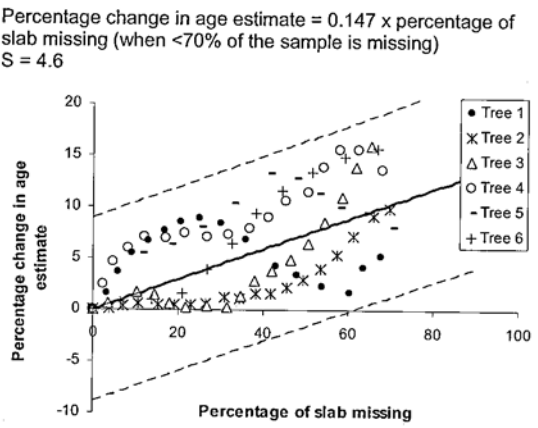


Figure 3.3 The absolute difference in age estimate, expressed as a percentage of the age of the tree, in relation to the percentage of the sample that was removed. The points are the data used to create the model and the lines are the mean and 95% confidence interval of the predictive model. S is the standard deviation of the model. A maximum of 70% of the sample was removed because beyond this point there was a marked change in error.

Examining the fit of the ring counts and linear extrapolation

The error of the RCAE method was estimated to be approximately 10% of the estimated tree age (Figure 3.4). The maximum error rate when using the RCAE method for the trees examined was ± 80 years. However, for most trees (88.1%) the error was ± 30 years or less. Of the trees examined, 25 were estimated to be older than 400 years using the RCAE Method. Although it is a generalisation and therefore exceptions do occur, many eucalypts are believed to reach only about 400 years of age (Rayner, 1992; Bradshaw and Rayner, 1997a). Fourteen of these older trees examined in the present study did not include 400 years within the error margins. The main reason for the potential increase in error for these trees is believed to be due to butt swell of the trunk, which was found in 13 of the 25 trees. When butt swell occurs there is often compression and expansion in the wood creating difficulties in tree ring identification. Other reasons found for the potentially excessive error in these trees was that a very small proportion of the radius was examined (four trees) or the rings were especially difficult to count (three trees). For three trees no reason for excessive error could be found and for two trees the size and condition of the stump suggested that the ages were appropriate (415 ± 35 years and 438 ± 36 years).

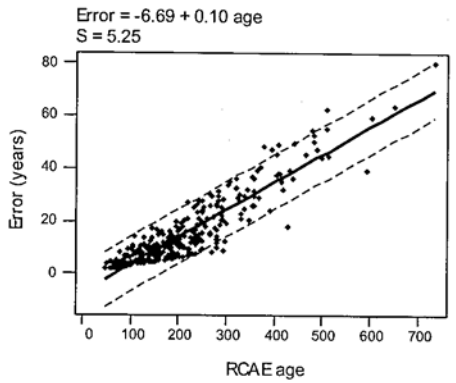


Figure 3.4 The error (in years) of each tree age estimate in relation to the RCAE (Ring Count And Extrapolation) age of the tree (n = 329). The line is the result of the regression between the RCAE age and error in the estimate. The dashed lines are the 95% confidence intervals of the data. The equation for the solid line is displayed above. S is the standard deviation in the data.

Growth models and associated error

Modelling the influence of site and tree variables on growth rates

When using site and tree variables, the best model predicting the age of the tree included a 2nd degree polynomial relationship with tree diameter, a negative relationship with soil phosphorus, average temperature, stand age classification and aspect (Table 3.4). Site aspects north, east and south were combined due to similarity in the data distribution, resulting in an

improved fit of the model. The stand age categories 'regrowth' and 'regrowth with mature' were also combined into only one category, referred to in Table 3.4 as simply 'Regrowth'. The polynomial relationship with tree diameter means there is a very slight tendency for tree growth rates to slow down with increasing size. The negative relationships with the other variables means that trees are younger for the same size, that is they have faster growth rates. When the random factor 'site' was included there was a decrease in DIC value of 24 units indicating there was some unexplained variation between sites. However, as the model is to be used for predictive purposes, the coefficients are indicated for a model excluding the random factor. The WinBUGS code for this model is provided in Appendix 2. Examination of the residuals indicated that trees older than 300 years were more likely to be underestimated in age when using the predictive model. The average absolute residual was 48.2 ± 44.3 s.d., indicating low levels of accuracy.

The variable 'forest type' was not selected as being an important influential variable in the Predictive Model. This is despite the fact that there was a significant difference in growth rate between the different forest types ($F = 7.123$, $p = 0.001$, $df = 2$), although the difference was only significant between dry forest and the others and there was a large amount of overlap between the different forest types (Figure 3.5). The mean and standard deviation of the Increment Formulas was 0.449 ± 0.16 for dry forest, 0.376 ± 0.15 for damp forest and 0.392 ± 0.16 for wet forest. If these values are used to estimate the age of a tree approximately 100 cm in diameter (i.e. 500 mm in radius under bark), the ages produced are: 224.5 ± 80 years for dry forest, 188 ± 75 years for damp forest and 196 ± 80 years for wet forest.

Table 3.4 Coefficients (mean, standard deviation, 2.5th and 97.5th percentile) of the explanatory variables included in the best Bayesian linear model for predicting tree age.

Variable	mean	s.d.	2.5%	97.5%	DIC[a]
Intercept	15.050	0.214	14.630	15.470	1758.9
Dbh	3.644	0.192	3.268	4.019	1529.7
Dbh^2	-0.463	0.064	-0.587	-0.337	1491.9
Mature	0				1465.7
Mature with regrowth	-1.091	0.338	-1.757	-0.428	
Regrowth[b]	-1.529	0.304	-2.126	-0.932	
Soil Phosphorus	-0.903	0.146	-1.190	-0.618	1448.7
Average temp	-0.623	0.141	-0.901	-0.346	1433.5
Aspect N, E, S	0				1429.5
Aspect W	-1.006	0.418	-1.827	-0.186	

[a] DIC values are the Bayesian equivalent of an AIC value to indicate progressive model improvement by addition of each new variable.

[b] Regrowth includes the stand age categories regrowth with mature and regrowth.

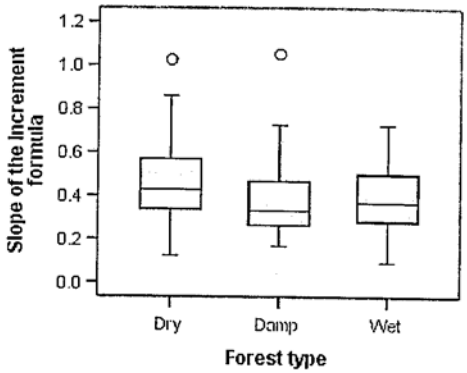


Figure 3.5 The relationship between tree growth rates (the slope of the Increment Formula) and forest type. The box length is the interquartile range. The circles are outliers with values between 1.5 and 3 box lengths from the upper or lower edge of the box.

Examining alternate growth models

Of the three growth models examined, the Site Formulas proved to have the best fit. This was evident in several ways. Firstly, a higher percentage of trees had the RCAE age lying within one standard deviation of the predicted age when using the Site Formulas than for either of the two other models (Table 3.5). Secondly, the regression of predicted tree age against RCAE tree age had a greater R^2 value for the Site Formulas. Finally, the relationship between the RCAE and predicted age had a slope coefficient closer to one than for either the Predictive Model or Overall Formula (Figure 3.6). The slopes of the Site Formulas ranged from 0.99 to 3.07. However, the error of the predictions varied greatly between sites (Figure 3.7b). The variability in the error terms was related to both the number of samples obtained at a site and the strength and linearity of the relationship between tree diameter and age found at a site. The site where the greatest error margins were found (see outliers in Figure 3.7b) contained two trees that were estimated to be around 600 years old and four trees that were just under half their size but estimated to be about 65 years old. As a consequence, the fit of the line was especially poor, which resulted in the large error estimates. Therefore, the Site Formulas provide accurate and precise estimates of tree age, but only when sufficient samples are used to construct the formula and when a strong linear association between tree age and diameter is evident at the site.

When there is no information on growth rates available at a site, a choice between the Predictive Model and the Overall Formula needs to be made. The Overall Formula had a greater percentage of trees whose RCAE age lay within one standard deviation of the predicted age than the Predictive Model. However, the R^2 of the association between RCAE age and predicted age was lower indicating a looser association (Table 3.5), as is evident in Figure 3.6. In addition, the slope coefficient of the relationship between RCAE and predicted

age was lower than was found for the Predictive Model indicating inferior accuracy in predictions. The reason for these apparently contradictory results is evident in Figure 3.7c. Given the Overall Formula is calculated by regressing RCAE age against tree diameter for all trees combined, a constant error rate of about 80 years is predicted across all tree ages (Figure 3.7c). This error margin is larger than that of the Predictive Model until trees reach about 300 years. Having larger error margins mean that the RCAE age has a greater likelihood of lying within one standard deviation of the predicted age. This indicates there is a lack of precision in the results which is undesirable. It therefore seems that when no site specific information is available, the Predictive Model is the superior choice. The predictions made by the Predictive Model are relatively well correlated with RCAE age (Table 3.5, Figure 3.6a) and the error margins are more flexible and reasonable, increasing with estimated tree age (Figure 3.7a). The relationship between tree age and error for the Predictive Model is: Error = 0.152 × age + 28.8.

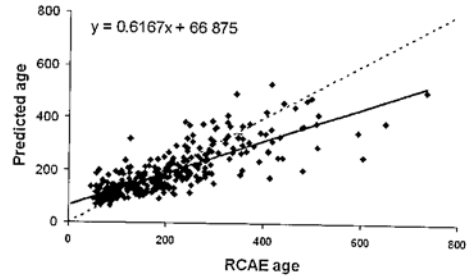
Table 3.5 Comparing the accuracy of the growth models.

Method	Percentage of trees within one standard deviation [a]	Adjusted R^2 of regression with RCAE age [b]
Predictive Model	67.8	0.642
Site Formulas	84.2	0.699
Overall Formula	74.5	0.426

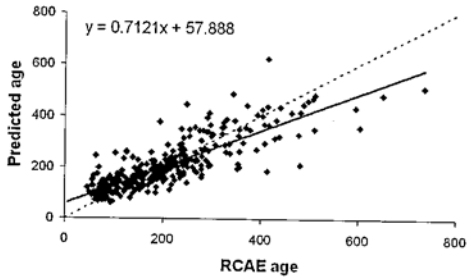
[a] The percentage of trees whose RCAE (Ring Count And Extrapolation) age was found to lie within one standard deviation of the predicted age for the three growth models considered (n = 329).

[b] The adjusted R^2 value of the regression analysis between the predicted results from the growth models and the RCAE age.

a) Predictive Model



b) Site Formulas



c) Overall Formula

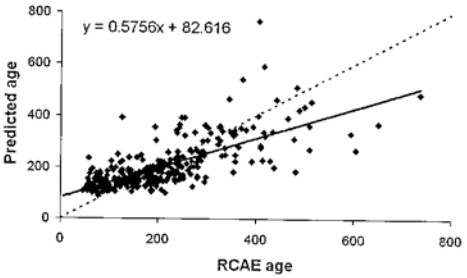
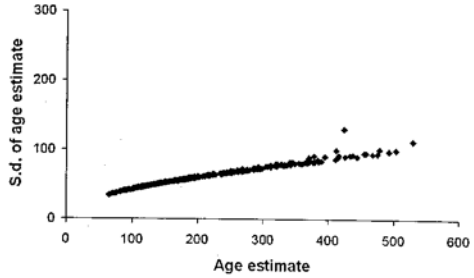
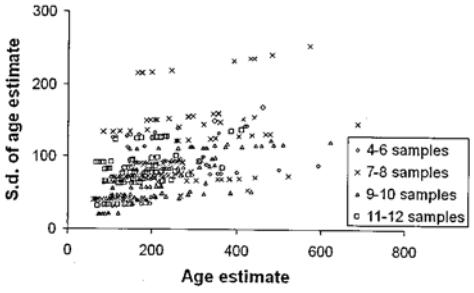


Figure 3.6 The predicted age of each tree plotted against the RCAE (Ring Count And Extrapolation) age of that tree using (a) the Predictive Model, (b) the Site Formulas and (c) the Overall Formula. The solid line is the result of the regression between the RCAE age and predicted age. The dashed line has a slope value of one and indicates perfect correspondence between predicted and RCAE age.

a) Predictive Model



b) Site Formulas



c) Overall Formula

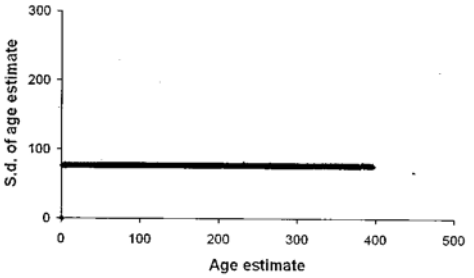


Figure 3.7 The standard deviation of the age estimate in relation to estimated age for all trees when using (a) the Predictive Model, (b) the Site Formulas and (c) the Overall Formula. The different symbols in Figure b indicate the number of wood samples that were used to establish the Site Formula.

Discussion

Of the two main methods for ageing trees examined in this study, ring counting and growth models, ring counting is the more accurate technique. There are two main sources of error associated with the ring counting method. There is the error found in the ring counting itself, and there is the error from extrapolating the missing section of wood. Only the second source of error could be examined in this study due to a lack of precise information on disturbance history. Figure 3.3 indicated that the amount of error resulting from extrapolating was approximately 15% of the proportion of the wood sample that was missing. This held true until approximately 70% of the wood sample was removed when the error rates showed a less consistent pattern. To estimate the total error resulting from this ageing technique the results from Brookhouse (1997) were used to estimate the error from the ring counts. The error from the extrapolation was then added to the ring counting error. The total error estimated to occur increased linearly with estimated tree age at a rate of roughly 10%, as seen in Figure 3.4.

The error rates of ring counting found in this study are larger than error rates reported in other studies. While some studies do not state how the error rates were obtained and are therefore of uncertain reliability (± 5%: White, 1971), others provide convincing evidence that accurate ages can be obtained (Banks, 1990; Brookhouse, 1997). Banks (1990) used cross-dating for several eucalypt species and found that the greatest inaccuracy of ring counting was only a few years. Banks (1993) cross-checked radii and estimated the error of ring counting in *Eucalyptus regnans* from the central highlands of Victoria to be two rings per 100. However, variability in accuracy has been found in some studies. Banks (1982) examined the rings of snow gums in alpine NSW and found that they were mostly accurate but some large discrepancies did occur. For example, ring counts ranged between 24 and 50 on one tree due to some compression in the wood. Banks (1997) also found variable accuracy when using radiocarbon dating to check ring counts. Two trees estimated by ring counting to be 170 and 130 years old showed reasonable accuracy, with radiocarbon dating estimating the trees to be between 116 and 176 years old. However, for one tree the radiocarbon dated age was double the estimated ring count age due to the indistinct outer rings on the tree (ring count age: 160; radiocarbon dated age 354 ± 30). This suggests that although accurate estimates for most trees can be obtained, substantial error can occur for some trees. The reason the current study had greater error rates than most other published work was most likely because trees were randomly chosen and there was no selection based on ring clarity or lack of rot in the wood. In addition, the error in the present study was assessed using a different technique to most studies, the option of cross-validation of rings and ages not being available.

The estimates of tree age and associated error in the current study are believed to be appropriate for most trees, but the error may be underestimated for the older and larger trees for several reasons. Firstly, the trees examined by Brookhouse (1997) which were used to estimate the error in ring counting were all less than 100 cm in diameter. Many of the trees examined in this study were larger than 100 cm and it is uncertain if ring counting error changes with greater tree diameter. Secondly, a linear extrapolation was used to estimate the age of the missing wood sample, despite the knowledge that trees often decrease in growth rate and produce narrower rings as they age (Rayner, 1992; Banks, 1993; but see Bradshaw and Rayner, 1997b). Thirdly, when just a small amount of wood was available for ring counting, which occurred most often in older trees, only the outer rings could be used to determine the extrapolation slope (Increment Formula). As these outer rings were formed during a period of reduced growth, this can lead to an overestimation of the age of the older and larger trees. Fourthly, the model estimating the error from extrapolation was produced from only six trees. While they were relatively large trees (maximum 165 cm in diameter), they were smaller than the biggest trees used in this study. It is uncertain if the results from this model provide accurate estimates of error for larger trees. Finally, while it appeared that the method for estimating tree age achieved reasonable accuracy for most trees, excessive error occurred in trees exhibiting 'butt swell' (an enlarging at the base of a tree). Butt swell occurs mostly in older stands and the magnitude of the butt swell increases with stem size (Walshe, 2001). It is consequently found mostly on the older and larger trees. Therefore, while the ages and associated errors are assumed to be reasonable for most trees, it potentially underestimates the error in age estimates for the largest and oldest of trees.

Evidence of a slight decrease in growth rate for older and larger trees was seen in the Predictive Model, which explored the site and tree characteristics relating to tree growth rates. This was apparent from the binomial effect of tree diameter. The variables found to be related to tree growth rates in addition to tree diameter were all site level variables; soil phosphorus levels, average temperature of the site, stand age and site aspect. Trees were found to grow faster (i.e. they were younger for the same size) on more nutritious soils (high phosphorus concentrations), on warmer and western facing sites (which tend to receive more rain) and in younger stands. Although the broad classification of forest type was not included in the Predictive Model, it was found that the trees in dry forest grew significantly slower than trees in either damp or wet forest. This effect was expressed in the model by the inclusion of soil phosphorus levels. Higher soil phosphorus levels correspond to an increase in forest wetness. This effect means that trees found in dry forest will be, on average, older for the same size than trees found in either damp or wet forest (for a tree 100 cm in diameter a mean difference of 36.5 years is expected with damp forest and 28.5 years with wet forest).

However, there was a large degree of overlap in the Increment Formulas between forest types meaning that prediction of growth rates based purely on forest type should not be done.

The variables conspicuous by their absence from the Predictive Model were tree shape and crown class. Other studies have shown that more senescent trees have slower growth rates than younger, healthier trees (Banks, 1993; but see Bradshaw and Rayner, 1997b). In addition, subdominant trees are less able to compete for sunlight and so are likely to have low growth rates. The lack of a relationship with tree dominance may be because either the data contained few suppressed trees or that some of those classified as suppressed were merely young and had not reached the canopy. However, up to 50% of annual rings can be 'missing' in suppressed trees (e.g. regrowth *Eucalyptus diversicolor*: Rayner, 1992), which means that these trees may have been older than they appeared. Therefore, the age of subdominant trees may have been less accurate than estimated despite the lack of a significant influence found in the current work. In relation to tree senescence, the relationship between tree senescence and stand age was not straightforward (33.1% of trees examined had dead tops in mature forest, 20.0% in mature with regrowth, 10.9% in regrowth with mature and 36.8% in regrowth forest) meaning that stand age is not an obvious surrogate for tree senescence in the current study. However, although tree shape did not contribute significantly to the growth model, it was significantly related to the slope of the Increment Formula. This suggests that the linear extrapolation is not entirely appropriate for these older trees, (although no single other transformation was appropriate either) and there may be an undetected decrease in growth rate with increasing senescence.

The variables which were found to influence tree growth rates were used to construct one of the three growth models examined in this study (the Predictive Model). The other two growth models were based on a regression between tree age and diameter, either using only trees at the site in question (Site Formulas), or when using all trees from all sites (Overall Formula). Upon examining the predictive ability and the error rates of the three growth models, the Site Formula proved to be the best for predictive purposes. The ring count and extrapolation (RCAE) age of the tree more frequently lay within the error margins of the Site Formula than for the other two growth models. In addition, the regression between the RCAE age and age as predicted by the Site Formulas had a slope that was closer to one than for the other two models, indicating more accurate predictions. The main concern with using the Site Formulas was that sufficient trees needed to be used to create the formula and greater error in predictions occurred when a non-linear association between tree age and diameter occurred. When it is not possible to use a Site Formula to estimate tree age, the Predictive Model proved superior to the Overall Model (a regression between tree diameter and age for all sites combined). However, the predictions made by all three growth models

had relatively large error bounds, indicating a lack of precision in predictions. It is therefore recommended that such growth models should only be used as a rough guide when no alternative is available. It should also be noted that these models need to be tested against new data in order to make a true assessment of their utility and accuracy. Gibbons *et al.* (2000b) tested the accuracy of their growth model by ring counting additional trees and found that in all but three of the 19 cases there was congruence between the age estimates.

Tree hollow management implications

While some studies have shown that eucalypts can form hollows at a very young age, most agree that more than 100 years is required (Ambrose, 1982; Gibbons *et al.*, 2000b; Whitford, 2002). However, eucalypts are unlikely to have hollows suitable for use by fauna if they are less than 120-180 years old, with large hollows being rare in eucalypts less than 220 years old (Gibbons and Lindenmayer, 2002). In terms of the tree species examined in this study, Ambrose (1982) estimated that *E. obliqua* in Victoria commenced hollow formation at 110 years and the maximum number of cavities occurs at 430 years. In south-eastern Australia, Gibbons *et al.* (2000b) found that less than 50% of *E. obliqua* and *E. fastigata* trees younger than 180 years old had hollows.

Although the rate of hollow occurrence increases with tree age (Mackowski, 1987; Whitford, 2002), the present study has shown that the error of the age estimate when ring counting also increases with tree age. (The error was less than ± 15 years for 72.8% of trees that were estimated to be between 100 and 300 years of age compared to an average of ± 42.4 years for trees >350 years old). However, the higher probability of having a hollow suitable for fauna in the older trees means that making management decisions is possible regardless of imprecision in tree age estimates. Greater accuracy is achieved for the age at which eucalypts generally being to produce hollows (100-200 years: Mackowski, 1987; Whitford, 2002). Establishing the age at which eucalypts produce hollows suitable for use by fauna is an important consideration when designing management strategies. The imprecision in age estimates is likely to be small enough for the younger trees to mean that age estimates are useful when making management decisions.

Greater error is generally found when using growth models compared with ring counting, but growth models will often be more useful in a management situation due to ease of application and their non-destructive nature. As the error of the age predictions can vary so greatly, it is difficult to make concrete recommendations. Therefore it is suggested that for age estimates using growth models, decisions based on tree-age data be weighted according to the lower error bounds of the age estimates. This precautionary approach would minimise the risk that the expected tree hollows are absent due to inaccuracy in tree age estimation.

Chapter 4

Errors associated with two methods of assessing tree hollow occurrence in *Eucalyptus obliqua* forest, Tasmania

Abstract

Tree hollows provide important habitat for fauna, but difficulties associated with detecting tree hollows can impede the formulation of appropriate management action. This paper examines the accuracy and errors associated with two methods commonly used to assess the presence/absence or abundance of tree hollows; 'ground-based' surveys and 'tree-felling' surveys. Three hundred and forty-six trees in Tasmania's State Forest were surveyed for hollows both before and after being felled. In order to assess the type and frequency of errors associated with each method, the fate of every potential hollow identified during the ground-based survey was determined after the tree had been felled. Three main types of error were identified: hollows misidentified during ground-based surveys, hollows not detected during ground-based surveys and hollows not found during tree-felling surveys. Bayesian models were used to examine the association between site, tree and hollow variables and the occurrence of these errors. The results showed that smaller hollows were more likely to be misidentified during ground-based surveys than larger hollows. In addition, larger trees were more likely to have hollows undetected by both survey techniques. Although ground-based surveys provide inaccurate hollow counts, they are useful for assessing relative rather than true hollow abundance, which is valuable in the selection of habitat trees for retention in production forests. Tree-felling surveys provide a more accurate measure of actual hollow counts, particularly when a correction is made for the proportion of tree that cannot be surveyed. Tree-felling surveys are therefore useful when constructing models predicting hollow abundance.

Introduction

Cavities in trees, tree hollows, provide important habitat for a range of fauna throughout the world (Lindenmayer *et al.*, 1996a; Webb and Shine, 1997; Martin *et al.*, 2004; Ruczynski and Bogdanowicz, 2005; Walker *et al.*, 2005). Many hollow-using animals are considered threatened, which is often at least partially attributed to a lack of suitable nesting sites (Smith *et al.*, 1985; Walker *et al.*, 2005; Monterrubio-Rico and Escalante-Pliego, 2006). A decrease

in the availability of hollow-bearing trees has been related to a range of factors including tree attrition, land clearance and silvicultural treatments (Gibbons and Lindenmayer, 2002).

In managed production forests, hollow-bearing trees are often retained to provide habitat for hollow-using fauna (Healy *et al.*, 1989; Wayne *et al.*, 2006). In order to ensure the tree retention guidelines provided are appropriate, information is required on the availability of hollows in different forest types, the proportion required to maintain populations of hollow-using fauna and the types of trees likely to provide habitat. However, hollow formation largely occurs by stochastic processes in hardwood forests and identification and measurement of tree hollows can be difficult. The difficulties associated with identifying tree hollows can impede the formulation of appropriate management guidelines (Healy *et al.*, 1989).

There are three methods that are commonly used to assess the availability of hollows: searching trees from the ground with the use of binoculars (ground-based or pre-fall surveys); climbing trees; and searching trees felled during a logging operation (tree-felling or post-fall surveys) (Table 4.1).

Ground-based surveys are quick and cheap, making them a useful and practical way to assess the presence of hollows in a tree (Table 4.1). However, this method does not provide accurate counts of hollow abundance (Healy *et al.*, 1989; Harper *et al.*, 2004). Viewing hollows from a distance can be difficult, especially under low light conditions or when obscured by foliage. Fire damage to the bark, sap stains or other marks can be mistakenly identified as being a hollow. In addition, most definitions of a hollow include a minimum depth (Whitford, 2002; Harper *et al.*, 2004) which cannot be accurately determined from the ground. This means that not all cavities identified during ground-based surveys will meet the definition of being a true hollow. As well as providing inaccurate counts of hollow abundance, ground-based surveys provide incomplete or inaccurate information on hollow dimensions and use of hollows by fauna (Gibbons and Lindenmayer, 2002).

Climbing trees to search for hollows, either with ropes or ladders, results in a large improvement in data quality, potentially providing accurate information on the occurrence and characteristics of hollows in a tree and the use of the tree by fauna (Haseler and Taylor, 1993; Harper *et al.*, 2004; Martin *et al.*, 2004). However, when using ladders only small trees or hollows lower down the tree can be examined (Martin *et al.*, 2004). The use of ropes requires specialised skills and is extremely time-consuming, thereby limiting the number of replicates that can be obtained within a particular budget (Table 4.1).

Tree-felling surveys can be done relatively quickly, thereby allowing larger sample sizes than tree-climbing surveys. They also allow collection of detailed information on the presence/absence, abundance and characteristics of hollows and their use by fauna. The largest samples of the internal dimensions of hollows, which are related to their use by fauna, have been obtained from tree-felling surveys (Gibbons and Lindenmayer, 2002). The disadvantages of this method include the fact that sampling is biased towards areas that have been approved for logging (Gibbons and Lindenmayer, 2002), that the technique cannot be used to identify habitat trees for retention in production landscapes due to the destructive nature of the sampling and that sections of some trees smash on felling or are obscured by the ground or logging debris (Table 4.1). The degree to which trees smash on felling depends on a variety of factors, including the amount of rot or dead wood they contain, the steepness of the slope on which they are felled and their height.

Table 4.1 A summary of the advantages and disadvantages of three methods for assessing the hollow resource.

Survey method	Advantages	Disadvantages	Example references
Ground-based survey	Quick Cheap Useful for determining hollow presence Potentially useful for determining hollow density Usable for all land tenures	Inaccurate Difficult in dense forest Provides limited data on hollow attributes and use	Lindenmayer *et al.*, 2000 Munks *et al.*, in press Walter and Maguire, 2005 Chambers and Mast, 2005
Tree-climbing survey	Provides accurate hollow counts Potentially provides information on hollow use Usable for most trees and forest types	Time consuming Expensive Requires specialist skills Produces smaller sample sizes for the same effort	Harper *et al.*, 2004 Haseler and Taylor, 1993 Ruczynski and Bogdanowicz, 2005
Tree-felling survey	Provides hollow count data Provides an assessment of hollow use Relatively quick and cheap so provides large sample sizes	Requires trees to be felled (which limits study location and ability to select habitat trees) Provides limited accuracy due to trees smashing or hollows being obscured	Gibbons *et al.*, 2000b Whitford, 2002 Mackowski, 1987

Of these three survey techniques, the one most often used by forest managers to assess the hollow resource is ground-based surveys. Such surveys are a good way to rapidly identify hollow-bearing trees for retention in production forests. Yet because hollows may be falsely identified or missed completely during a ground-based survey (Healy *et al.*, 1989) it is important to understand the accuracy of this technique. High error rates may mean the availability of hollows is overestimated or underestimated. Unsuitable trees may be retained

for animal habitat in production forests if hollow presence is overestimated. There have been some attempts to assess the accuracy of ground-based surveys (Healy *et al.*, 1989; Whitford, 2002; Harper *et al.*, 2004). This has been done by comparing the number of hollows seen during a ground-based survey and those recorded using an alternative method such as tree-climbing. These studies have generally acknowledged that ground-based surveys provide inaccurate estimates of hollow abundance, but they varied in how useful they perceived this survey method as being (Whitford, 2002; Harper *et al.*, 2004).

Correlating the results of two different methods is a sensible strategy when the alternative survey method provides accurate counts, as occurs with tree-climbing. However, when the alternative method is also inaccurate, as results from tree-felling surveys will be, it may be difficult to draw reasonable conclusions. To avoid the issue of inaccuracy in tree-felling surveys some authors assume that error rates (the degree to which trees are smashed or obscured) are constant for all trees (Gibbons, 1999). Assuming a constant rate of error allows the results of tree-felling surveys to be compared to ground-based surveys. However, trees often smash to different degrees meaning that constant error rates between trees will not always occur. When felled trees vary in the amount they smash it is necessary to examine the causes of error in more detail. An examination of the error may allow corrections to be made to estimates of hollow abundance.

One of the only studies found in the literature to assess the rate of hollow loss due to tree felling was done in *Eucalyptus pilularis* forest in NSW, Australia (Mackowski, 1987). It was found that the rate of hollow loss varied between tree size classes. However, this study was done by comparing counts of 'branch', 'stem' and 'trunk top' hollows before and after felling. This technique is simplistic and does not take into consideration the error associated with both of the survey methods. Furthermore, no assessment was made of the types of hollows that were lost during felling.

Animals can be selective in their use of tree hollows, preferentially using hollows of a particular size, shape and orientation (Saunders *et al.*, 1982; Inions *et al.*, 1989; Haseler and Taylor, 1993). In order to assess the importance of the error in tree hollow surveys and the implications this error has on fauna management, an examination is required of the type of hollows that are missed. If hollows that are generally unfavoured by animals are more likely to be missed during a survey, then the inaccuracy of hollow surveys is unimportant. However, if the hollows commonly missed are those preferred by animals, then the issue of survey error is of greater import. Only two references were found in the literature referring to the types of hollows that are missed during tree hollow surveys. Mackowski (1987) found that branch hollows are more likely to be missed than trunk or top hollows during ground-

based surveys. Whitford (2002) gave a personal observation that vertically oriented hollows are difficult to see from the ground.

This paper evaluates the accuracy of ground-based surveys and tree-felling surveys in wet and dry *Eucalyptus obliqua* forest in Tasmania. Studying the fate of each hollow allowed a more detailed assessment of the errors associated with these survey methods than has been undertaken in previous studies. Two key questions are explored: how accurate are these methods in identifying the presence/absence and abundance of tree hollows; and which hollows are likely to be missed or mistakenly identified? A cheap and simple method is proposed to improve the accuracy of hollow counts obtained during tree-felling surveys.

Methods

Study sites

Forestry coupes within State Forest in Tasmania, planned for harvest between January 2004 and May 2005, were selected in consultation with forest managers. Twenty-three coupes in dry *Eucalyptus obliqua* forest and eleven in wet *E. obliqua* forest were used as study sites (Figure 4.1). These broad forest types are two of the most widespread in Tasmania of use to the forest industry. What is referred to as 'damp' forest in the current paper (eight sites) is actually a subset of the greater classification 'dry'. However, as trees in this classification showed characteristics of both wet and dry *E. obliqua* forest it was decided to give the subset a separate category (Table 4.2). Ground-based and tree-felling surveys will be referred to as pre-fall and post-fall surveys respectively throughout the methods and results of this paper to ensure clarification of the survey method in question.

Figure 4.1 Location of the study sites in Tasmania, Australia. Squares indicate dry forest, triangles indicate damp forest and circles indicate wet forest.

Pre-fall surveys

Approximately thirteen trees with a diameter at breast height over bark (dbh) of 50 cm or greater were selected (the first haphazardly and the remaining being the closest tree that was further than 20 m from the previous) at each site and assessed for tree hollows. Each tree was surveyed for potential hollows from the ground using binoculars, with attempts made to view the tree on all sides. All observations were made by the author to prevent observer bias (Harper *et al.*, 2004). A hollow was defined as a cavity in the tree with a minimum entrance diameter and depth of two centimetres, located at least two metres above the ground. All potential hollows were classified by location (trunk or branch) and position (base, middle or end). A detailed description of the shape and location of each potential hollow was made to allow relocation and further examination after the tree had been felled.

Post-fall surveys

Trees selected during the pre-fall surveys were felled by forest contractors as part of normal forestry operations within the logging coupe. Not all trees surveyed in the pre-fall survey could be studied post-felling as some were not felled due to logistical constraints. A total of 346 trees were successfully surveyed post-felling, between six and twelve trees at a site (average 10 trees ± 1.3 s.d.), with an average diameter of 106.7 cm (± 50.2 s.d.). After felling, each of these trees was located and searched for hollows. For each tree, the proportion of both the trunk and the canopy that could not be surveyed was estimated. The location of each potential hollow noted in the pre-fall survey was then found (if possible) and allocated to one of three categories:

- 'True hollow' (minimum depth and entrance diameter of 2 cm);
- 'Lost hollow' (hollow was not found, was obscured by the ground or was too smashed to measure);
- 'False hollow' (hollow was identified in the pre-fall survey but on the post-fall survey was found to not fit the classification of a hollow).

False hollows were further categorised according to the reason for misidentification:

- Shallow (the hollow was < 2 cm deep);
- Small (the minimum entrance diameter was < 2 cm wide);
- Indent (the misidentified hollow was not a cavity in the wood but a hole in the bark);
- Mark (there was a mark on the tree which was mistaken for a hollow);
- Unknown (the location of the misidentified hollow was established but no sign of a hollow could be found).

Hollows found in the post-fall survey that were not detected during the pre-fall survey are referred to as 'New hollows'.

Site variables, tree and hollow characteristics

Information on a number of habitat variables, tree and hollow characteristics were collected during both the pre-fall and post-fall surveys (Table 4.2). These variables were chosen for their anticipated value as predictors of the rate of error for both survey methods. For four of the 34 sites, no basal area measurement was done. Of the 346 trees assessed, no estimate of trunk access or crown access was made for one tree. Not all hollow attributes could be measured for each hollow due to the destructive nature of the sampling. Adjustments to sample sizes are consequently stated, where required, throughout the text.

Data analysis

False hollows

The rates at which hollows were mistakenly identified (False hollows) and reasons why they were incorrectly identified during the pre-fall surveys were explored using chi-square analysis. As some potential hollows noted in the pre-fall survey were not found during the post-fall survey (Lost hollows), the counts of False hollows were compared to the counts of True hollows. Bayesian Bernoulli mixed-effect models with uninformative priors in WinBUGS 1.4 (Spiegelhalter *et al.*, 2003), were used to identify the most important variables related to whether a hollow was mistakenly identified (False) or that a tree contained a False hollow. Model selection was done using a forward stepwise approach investigating single and two-way interactions. The random variables tree and site number were used where appropriate. The predictor variables are outlined in Table 4.2. The 'pre' version of min hollow size (see Table 4.2) was used for the hollow level model as entrance dimensions could not be measured on False hollows in the post-fall survey. All continuous variables were centred (mean subtracted from the value: McCarthy, 2007) to reduce autocorrelation between successive samples. The initial 1,000 samples were discarded as a 'burn in' and the following 10,000 samples were used to calculate the Deviance Information Criterion (DIC), after establishing that the DIC value stabilised after this many samples. DIC values can be interpreted in a similar way to AIC values (Burnham and Anderson, 2002; Spiegelhalter *et al.*, 2002) and were used for model selection. A difference in DIC value of less than two indicates a lack of difference in the models, while a difference of three of more indicates that the model with the smaller DIC value is superior (McCarthy, 2007). After selection of the final model, 100,000 samples were taken to establish model parameters. The mean, standard deviation of the model coefficients and the 2.5^{th} and 97.5^{th} percentiles of the distribution for the final model were calculated. These percentiles represent a 95% Bayesian confidence interval. To evaluate the fit of the model all data used to produce the model were used to predict whether a tree would contain a False hollow or not, or whether a hollow would be 'False' or not. The number of informative predictions was examined where a

prediction was considered to be informative if the 95% credible interval did not span from zero to one. In addition, the fit was assessed by the following formula which estimates the proportion of deviance explained by the fitted model:

$$GOF = (Dhat_1 - Dhat_0) / Dhat_0$$

GOF = Goodness of fit measure.

$Dhat_1$ is the Dhat value produced by WinBUGS for the fitted model (the Dhat value represents the deviance at the mean of the posterior distribution: McCarthy, 2007).

$Dhat_0$ is the Dhat value where all parameters are removed except an uninformative constant and an uninformative dispersion parameter (where it is included in the fitted model).

New hollows

Bayesian Poisson mixed-effect modelling, including the random variable 'site', was used to examine the relationship between the occurrence of 'New hollows' and the site and tree variables. The dependent variable was the square-root of the number of New hollows found on a felled tree and the predictor variables are described Table 4.2. All data used to produce the model were then used to make predictions in order to test the fit of the model. The predicted values were subtracted from the observed values to produce the residuals, which were graphically examined to assess the fit of the model. The GOF value was also calculated. The variables relating to whether a particular hollow was observed in the pre-fall survey or not were examined using Bayesian Bernoulli mixed-effect modelling, including the random variables 'tree' and 'site'. The 'post' version of the variable min hollow size (Table 4.2) was used in the modelling process as hollows classified as 'New' were not observed in the pre-fall survey and therefore no estimates of entrance parameters were made pre-felling. The number of informative predictions and the GOF value were examined to assess the fit of the model.

Lost Hollows

The most important predictor variables related to the number of Lost hollows on a tree was assessed using Bayesian Poisson mixed-effects modelling, including the random variable 'site'. Bayesian Bernoulli mixed-effect modelling was used, including the random variables 'tree' and 'site', to determine the most important site, tree and hollow variables related to whether a particular hollow observed during a pre-fall survey was lost or not. The 'pre' version of min hollow size was used (Table 4.2). The fit of the models were assessed as for those above. In addition, the number of branch and trunk hollows that were 'lost' were regressed against the proportion of canopy and the proportion of trunk that was unobserved respectively to determine if they were related.

Comparing pre-fall and post-fall hollow counts

Hollow counts recorded during the pre-fall survey were compared with hollow counts recorded during the post-fall survey using Pearsons' correlation in the program SPSS (2005). Pre-fall counts were also compared with corrected post-fall counts, which took into account the hollows present on the parts of the tree that were obscured or had been destroyed during felling. It was assumed that the distribution of hollows was constant across the trunk and canopy of the tree. These 'corrected' post-fall counts were estimated by dividing the number of hollows observed on the trunk by the proportion of the trunk that was successfully inspected. This was repeated for the canopy, and the results were summed to produce the total estimate of hollow abundance for the tree as follows:

$$H_c = (T / P_t) + (C / P_c)$$

Where:

H_c is the corrected number of hollows estimated to actually exist on the tree

T is the number of hollows counted on the trunk of the tree during the post-fall survey

C is the number of hollows counted on the canopy of the tree during the post-fall survey

P_t is the proportion of the trunk that was estimated to be successfully surveyed during the post-fall survey

P_c is the proportion of the canopy that was estimated to be successfully surveyed during the post-fall survey

To examine if the correction procedure outlined above was a suitable method for correcting hollow abundance estimates, the difference between the pre-fall counts and the corrected counts was examined. This difference was modelled as was done for the number of Lost hollows so that it could be observed how well the correction procedure mirrored the process of losing hollows. That is, Bayesian Poisson mixed-effect modelling was used to examine which site, tree and hollow variables (Table 4.2) were related to the difference in counts. The fit of this model was assessed as for the previous models. If the model predicting the difference in counts is similar to the model predicting the number of hollows that were lost, this indicates that the correction procedure approximates the rate at which hollows were lost during the felling of the tree.

Table 4.2 Site, tree and hollow attributes recorded during pre-fall and post-fall surveys.

Site, tree or hollow attribute	Description
Basal area	The basal area of each stem with a DBH > 10 cm was measured in a 0.25 ha circular plot. The actual radius used was dependent on the slope of the site to ensure a horizontal area of 0.25 ha. The results were combined and extrapolated to produce an estimate of the basal area per hectare at each site.
Vegetation layers	The number of distinct vegetation layers at each site.
Canopy cover	The percent cover of the canopy was estimated at each site using a modified Braun–Blanquet index (Mueller-Dombois and Ellenberg, 1974).
Understorey cover	The percent cover of the understorey was estimated at each site using a modified Braun–Blanquet index (Mueller-Dombois and Ellenberg, 1974).
Forest type	The type of forest at each site was classified as: Dry: understorey dominated by bracken or low shrubs; Damp: understorey dominated by shrubs over 2 m, often including broad-leaved species; Wet: understorey dominated by broad-leaved species.
Dbh	The diameter of the tree (cm) measured at 1.3 m using a diameter tape.
Pre-fall counts	The number of potential hollows observed during the pre-fall survey.
Tree shape	The shape of the tree was assigned to one of the following categories based on those proposed by Smith and Lindenmayer (1988): (3) tree with full round crown; (4) mature tree with large branches coming off the stem; (5) mature tree with numerous dead branches; (6) mature tree with dead top but big crown; (7) mature tree with dead top and small crown; (8) burnt tree with epicormic growth.
Crown class	Tree dominance was classified by assessing the height of the tree relative to those surrounding it: (1) dominant (emergent); (2) co-dominant; (3) subdominant or suppressed.
Tree visibility	An indication of the percentage of the tree that could be viewed from the ground: High: observed 2/3 or more of the tree; Med: observed between 1/3 and 2/3 of the tree; Low: observed less than 1/3 of the tree.
Burn damage	A measure of the intensity of damage done by fire to the tree: (1) no indication of fire damage; (2) some charcoal on bark; (3) burnt bark and scarring; (4) severely burnt, forming a bridge.
Fire scars	The number of fire scars seen on the trunk was counted.
Dead branch index	The number of dead branches in each of the following size categories was estimated: (S) small (5-15 cm); (M) medium (15-40 cm); (L) large (> 40 cm). An estimate of dead branch index was obtained by the following formula: DBI = S + 5M + 10L.
Canopy access	The quantity of vegetation in close proximity to the tree foliage based on (Lindenmayer *et al.*, 1996a). One point is scored for each 3 m vertical section of a hollow-bearing tree where surrounding vegetation was < 10 cm from the tree, on either side of the tree. A cumulative score is calculated.
Trunk access	As above but on the trunk.
Tree height	Tree height (m) was determined using a clinometer.
Height to crown	The height (m) to the base of the crown was determined using a clinometer.
Crown volume	The volume of the crown (m^3) as estimated using the diameter of the crown north-south and east-west and the estimated depth of the crown.

Table 4.2 continued

Site, tree or hollow attribute	Description
Min hollow size	The smallest entrance dimension of a hollow (cm). (Pre) indicates this version of the variable was estimated during the pre-fall survey and was used in the False and Lost hollow models. (Post) indicates the value was measured during the post-fall survey and was used in the New hollow model.
Location	Location of the observed hollow: trunk or branch.
Orient	Orientation of the observed hollow: up; horizontal; or down.
Hollow height	The height of the hollow above the ground as measured post-felling (m).

Results

During the pre-fall surveys, 1,348 potential hollows were observed in 272 of the 346 trees examined. During the post-fall surveys, 584 hollows in 210 trees were identified as 'True hollows', 101 hollows in 73 trees were found to be 'False hollows' and 663 hollows in 202 trees were 'Lost'. 1,523 'New hollows' were located during the post-felling surveys.

Hollow, tree and site characteristics and 'False Hollows'

The main reason why hollows were mistakenly identified during the pre-fall surveys was because they were less than two centimetres deep (Figure 4.2). There was no significant difference in the reasons why hollows were mistakenly identified between forest types when all forest types were included ($\chi^2 = 5.5$, $p = 0.701$, $df = 8$). However, 60% of the categories had an expected frequency of less than five, which is greater than the recommended 20% (Quinn and Keough, 2002), bringing the validity of the analysis into question. Consequently, the data was reanalysed after combining most of the categories, resulting in only two categories: 'shallow' and 'other'. While dry and damp forest had more shallow hollows than wet forest, the result was non-significant ($\chi^2 = 3.071$, $p = 0.215$, $df = 2$).

The best model explaining if a tree contained a False hollow or not included dead branch index and the log of the number of hollows seen in the pre-fall survey plus one (Table 4.3a). As the number of potential hollows observed in a tree during pre-fall surveys increased, the chance of one being mistakenly identified increased initially then gradually levelled off. The inclusion of dead branch index in the model indicates that the less dead wood a tree has, the more likely the tree is to have a False hollow. This is most likely to be because more dead wood is a sign of advanced senescence and hollows are less likely to be shallow in more senescent trees. Removal of the random factor 'site' caused a very slight improvement in the model indicating that sites did not differ in their likelihood of having mistakenly identified

hollows. In examining the predictive ability of the tree-level model, only eight of the 346 trees used for analysis were predicted to definitely not have a False hollow, the 95% credible intervals of the remaining trees spanning from zero to one. The GOF value was 0.12 indicating this model explained very little of the deviance in the data and therefore has poor predictive ability.

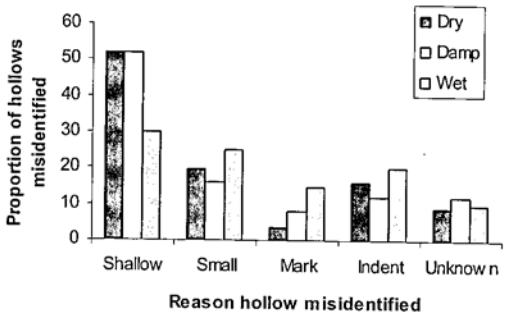


Figure 4.2 The reasons why hollows were misidentified according to forest type; n = 56 for dry forest, 25 for damp forest and 20 for wet forest. Shallow: the depth of the hole was < 2 cm; Small: the minimum entrance diameter was < 2 cm; Mark: there was a mark on the tree which was mistaken for a hollow; Indent: it was not a true cavity but only a hole in the bark; Unknown: the location was found but no evidence for a hollow could be seen.

Twenty-five of the hollows examined did not have a minimum hollow entrance width estimated during the pre-fall survey due to a lack of clear visibility. Of these, four were False hollows. Therefore the modelling of whether a hollow was falsely identified or not was done using 97 False hollows and 563 True hollows, from 223 trees at 34 sites. The best model included the variables minimum hollow size and dead branch index. Both these variables had a negative effect on the likelihood of a hollow being mistakenly identified (Table 4.3b). That is, smaller hollows, especially those in trees with little dead wood, were less likely to be True hollows. Removal of the random variables site and tree did not affect the model meaning that there was no significant difference between sites or trees in the likelihood of incorrectly identifying a hollow. Examination of the model showed that none of the hollows examined were predicted to definitely not be a False hollow, the predictions for all hollows having a 95% credible interval spanning from zero to one. The GOF value was 0.15, indicating poor predictive ability.

Table 4.3 Coefficients (mean, standard deviation, 2.5th and 97.5th percentile) of the explanatory variables included in the best Bayesian regression model for whether (a) a tree contained a misidentified hollow or not and (b) a hollow was misidentified or not.

a)

Variable	mean	sd	2.5%	97.5%	DIC[a]
Intercept	-1.5970	0.1625	-1.9250	-1.289	358.6
Log (Pre-fall count +1)	1.4000	0.2331	0.9570	1.8700	334.8
Dead branch index	-0.0239	0.0062	-0.0365	-0.0124	318.2

b)

Variable	mean	sd	2.5%	97.5%	DIC
Intercept	-2.4540	0.1946	-2.4470	-2.094	553.0
Min hollow size (pre)	-0.1498	0.0319	-0.2156	-0.0908	503.0
Dead branch index	-0.0218	0.0046	-0.0312	-0.0134	473.3

[a] The DIC values are the progressive values attained with the addition of each new variable.

Hollow, tree and site characteristics and 'New hollows'

The best model explaining the number of New hollows found on a tree included the predictor variables pre-fall hollow counts and tree diameter (Table 4.4a). There were more New hollows found when trees were larger and when more hollows had been seen in the pre-fall survey. The variable site did not contribute significantly to the model indicating that there was no variation between sites. When examining the residual plots for the number of New hollows, there were two large outliers. The largest (residual difference –47) was for an unusually large tree. The average absolute difference between the observed and predicted data was 3.3 ± 4.4. The GOF value was 0.11 indicating that the model explained little of the deviance in the data and has poor predictive ability.

Not all attributes could be measured for the 2,107 hollows found in the post-fall survey, due to varying degrees of smashing. Consequently, only 1,816 hollows were used to model whether a particular hollow had been seen in the pre-fall survey or not. Of these, 526 had been seen in the pre-fall survey. The best model included location of the hollow, height of the hollow above the ground, the minimum size and aspect of the hollow. Branch hollows, small hollows, hollows high off the ground and those facing upwards were less likely to be seen during the pre-fall surveys. Hollows facing down were more likely to be seen during pre-fall surveys than those facing horizontally. The random variables, tree and site, both contributed significantly to the model meaning that there was unexplained variation between sites and trees (Table 4.4b). There were four trees that were outliers, either because they had an unusually high or low rate of observing hollows in the pre-fall survey. The model improved significantly when these trees were removed from the analysis and the random factor 'tree' no longer contributed significantly to the model, although site still did. The reason for these trees having different rates of hollow detection was unclear and their removal caused only minimal changes in the model values so the results are presented using

all the data. However, examination of the predictive ability of this model found that only one of the hollows examined had an informative distribution as predicted by this model (i.e. the 95% credible interval did not span from zero to one). The GOF value for this model was 0.24 indicating the model explained less than a quarter of the variation in the data, but a large proportion of this was explained by the random variables 'site' and 'tree' (GOF value without the random variables = 0.12). Therefore this model has very little predictive ability.

Table 4.4 Coefficients (mean, standard deviation, 2.5th and 97.5th percentile) of the explanatory variables included in the best Bayesian regression model for (a) the square-root of the number of 'New hollows' found on a tree and (b) whether a hollow was seen in the pre-fall survey or not.

a)

Variable	mean	sd	2.5%	97.5%	DIC[a]
Intercept	0.4037	0.04563	0.3130	0.4041	1121.3
Pre-fall count	0.2598	0.0381	0.1846	0.3343	1010.3
Tree diameter	0.1474	0.0395	0.0693	0.2242	999.0

b)

Variable	mean	sd	2.5%	97.5%	DIC
Intercept	-1.4130	0.2185	-1.8460	-0.9859	2187.9
Min hollow size (post)	0.0631	0.0081	0.0475	0.0793	2105.6
Location: Trunk	0				2015.3
Location: Branch	-0.7609	0.1462	-1.0490	-0.4756	
Hollow height	-0.4622	0.0818	-0.6234	-0.3032	1968.8
Orient - Up	0				1934.6
Orient - Horizontal	0.9906	0.1721	0.6584	1.3320	
Orient – Down	1.4320	0.5153	0.3955	2.4280	
Sdsite[b]	0.6166	0.1374	0.3710	0.9099	1864.2
Sdtree[c]	0.6099	0.1083	0.4021	0.8289	1835.8

[a] DIC values are the progressive values attained with the addition of each new variable.

[b] 'Sdsite' is the standard deviation of the random effect site.

[c] 'Sdtree' is the standard deviation of the random effect tree.

Hollow, tree and site characteristics and 'Lost hollows'

When using only a linear relationship with pre-fall hollow counts the best model determining the number of hollows lost on a tree included pre-fall counts, tree diameter and tree shape (DIC = 975.1). Increasing tree shape class (i.e. more senescent trees) resulted in more Lost hollows. However, the best model overall included tree diameter and a 2nd degree polynomial relationship with pre-fall hollow counts (DIC = 884.1, Table 4.5a). The random factor 'site' did not contribute significantly to the model meaning that the number of Lost hollows did not vary between sites. The GOF value was 0.48 indicating that the model explained almost half of the deviance in the data and has fair predictive ability. In addition, the number of hollows that were lost from the canopy was significantly related to the proportion of canopy that was not examined ($F = 37.7$, $p < 0.01$) and the number lost on the

trunk was related to the proportion of trunk that was not observed ($F = 22.1$, $p < 0.01$) (Figure 4.3).

When using all 1,348 hollows, the best model produced explaining whether a particular hollow observed in the pre-fall survey would be lost in the post-fall survey included whether the hollow was on the trunk or a branch and tree diameter (Table 4.5b). Branch hollows were more likely to be lost than trunk hollows and hollows were more likely to be lost on larger trees. The inclusion of dead branch index instead of tree diameter was an equally acceptable model. The random variable site did not contribute significantly to the model meaning that the likelihood of a particular hollow being lost did not vary between sites. However, removing the random variable 'tree' did significantly affect the fit of the model meaning that the likelihood of finding a hollow varied between trees according to unmeasured characteristics. Examination of the predictive ability of this model revealed that no hollows were given an informative distribution by the model, with all 95% credible intervals ranging from zero to one. The GOF value of this model was 0.15 indicating the model explains little of the deviance in the data. Furthermore, almost all of the explanatory power it does have comes from the random variable 'tree' (GOF without tree = 0.026). Therefore this model has no predictive ability.

Table 4.5 Coefficients (mean, standard deviation, 2.5th and 97.5th percentile) of the explanatory variables included in the best Bayesian regression model for (a) the number of hollows that were not found on a tree in the post-fall survey and (b) whether a particular hollow was found or not.

a)

Variable	mean	sd	2.5%	97.5%	DIC[a]
Intercept	0.3201	0.0569	0.2080	0.4309	1677.8
Tree diameter	0.0836	0.0323	0.0199	0.1466	1364.6
Pre-fall count	0.3171	0.0181	0.2819	0.3531	988.9
Pre-fall count^2	-0.0152	0.0016	-0.0183	-0.0122	884.1

b)

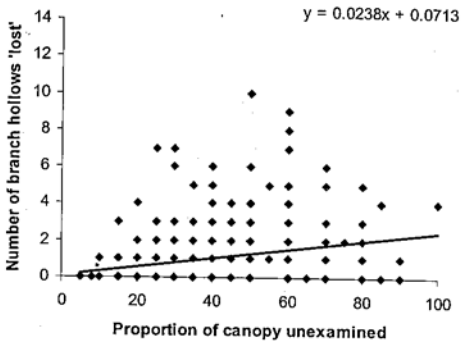
Variable	mean	sd	2.5%	97.5%	DIC
Intercept	-0.6102	0.1165	-0.8431	-0.3844	1870.4
Tree diameter	0.0064	0.0014	0.0037	0.0092	1849.7
Location: Trunk	0				1826.5
Location: Branch	0.6869	0.1324	0.4279	0.9472	
Sdtree[b]	0.7558	0.1086	0.5507	1.9763	1766.9

[a] DIC values are the progressive values attained with the addition of each new variable.

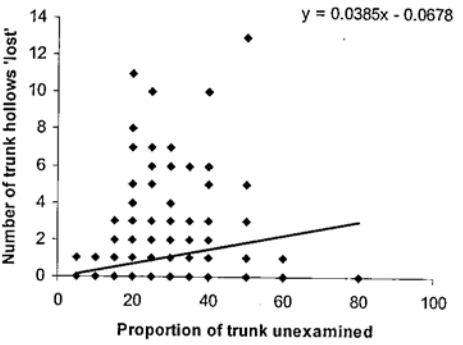
[b] 'Sdtree' is the standard deviation of the random variable 'tree'.

Figure 4.3 The relationship between the number of hollows seen in the pre-fall survey that were not found (lost) in the post-fall survey for (a) branch hollows and (b) trunk hollows.

a)



b)



Comparing pre-fall and post-fall hollow counts

All correlations between pre-fall and post-fall counts were highly significant. Overall, pre-fall hollow counts show a slightly stronger correlation with uncorrected post-fall counts than with the corrected post-fall counts. Hollows located on the trunk during the pre-fall survey were more strongly correlated to corrected post-fall counts than branch hollows, suggesting greater accuracy in detecting trunk hollows during pre-fall surveys (Table 4.6).

Table 4.6 Results of the Pearsons' Correlation analysis between pre-fall and post-fall hollow counts and pre-fall and corrected hollow counts (n = 343).[a]

	Corrected	Uncorrected
Trunk hollows	0.811	0.846
Branch hollows	0.663	0.668
Total hollows	0.750	0.787

[a] The results for all analyses had a p-value of < 0.001. The different sample size for this analysis is because for three trees no estimate of the amount of tree unsurveyed was made and so the corrected hollow count could not be made.

The model exploring the relationship between site and tree parameters and the difference in number of hollows between the pre-fall survey and the corrected hollow estimates was constructed using 343 trees. The smaller sample size resulted from three trees having no estimate of the amount of tree unsurveyed during the post-fall survey and so no corrected hollow counts could be made. The best model explaining the difference in number of hollows contained the parameters tree diameter and a 2nd degree polynomial relationship with pre-fall hollow counts (Table 4.7). This means that the accuracy of pre-fall survey hollow counts decreased both with increasing tree diameter and the number of hollows that were detected. The removal of the random variable 'site' resulted in a decrease in DIC value of five, which was considered low enough to justify removal. However the dispersion parameter did contribute significantly to the model and so was retained. The GOF for the model was 0.05, indicating that very little of the deviance was explained. Examination of the residual plot for the first model found that there was one extreme outlier (residual value of -105), which was the largest tree examined. There were six other trees that had residual values greater than twenty. However, 77% of the residuals have a value less than three indicating that the model had a relatively good fit for most trees. The predictive model overestimated the difference in hollow counts for 67.5% of the trees examined. It should be noted that the parameters in this model are exactly the same as those for the model predicting the number of hollows that were lost, apart from the inclusion of the dispersal parameter. The distributions of the coefficients for the two models overlap for all variables except the intercept and tree diameter. The intercept is greater in the 'difference' model and the coefficient for tree diameter is greater in the Lost model. The occurrence of the same variables and the overlap in the distribution of several of the coefficients indicates that the correction procedure approximates the rates at which hollows were lost.

Table 4.7 Coefficients (mean, standard deviation, 2.5th and 97.5th percentile) of the explanatory variables included in the best Bayesian mixed-effect model explaining the difference in pre-fall and corrected hollow counts.

Variable	mean	sd	2.5%	97.5%	DIC[a]
Intercept	0.8343	0.0755	0.6829	0.9783	3150.0
Disp[b]	0.7509	0.0533	0.6522	0.8603	1331.7
Tree diameter	0.0085	0.0012	0.0085	0.0109	1279.5
Pre-fall count	0.2692	0.0242	0.2225	0.3177	1261.7
Pre-fall count^2	-0.0167	0.0025	-0.0219	-0.0117	1234.1

[a] The DIC values are the progressive values attained with the addition of each new variable.

[b] 'Disp is the standard deviation of the over-dispersion parameter.

Discussion

This study found considerable inaccuracy in both of the survey methods examined, but the overall error for ground-based surveys was much higher than that for tree-felling surveys. The models explained only small amounts of the variability in the data (with the exception of the model predicting the number of hollows lost during the post-fall survey), meaning that there is considerable difficulty in predicting the occurrence of error. It is therefore recommended that measures be taken during surveys, whenever possible, to minimise the sources of error. Given that total elimination of error is unlikely, extrapolation or correction procedures may improve the accuracy of survey methods. Regardless, it is important to consider the implications of survey error on the utility of tree hollow survey methods to be sure the appropriate technique is used for a study.

Tree-felling surveys

Tree-felling, or post-fall surveys are often used to develop models predicting the occurrence of suitable habitat trees by collecting information on the presence and abundance of hollows and the use of trees by fauna (Gibbons, 1999). Larger, older trees with a number of observed hollows are generally thought to provide good habitat for fauna and are therefore more likely to be chosen as habitat trees (Wayne *et al.*, 2006). However, the present study found an increase in the number of hollows that were 'lost' or not found during the post-fall survey with increasing tree diameter and number of observable hollows. These two variables explained almost half of the variability in the data. This indicates that the larger, more damaged (and thus hollow-bearing) trees smash more on felling, meaning less sound tree is available for examination during the post-felling survey and more hollows are undetected (i.e. lost).

In the current study, the rate of hollow loss due to tree-felling was high, with 49.2% of hollows observed in the pre-fall survey not detected post-felling due to smashing or obscuring. The loss of a particular hollow was related to tree size, hollow abundance and whether the hollow was located in a branch or on the trunk. It was also found that there was a significant difference between trees in whether a hollow would be lost during tree felling. This most likely reflects the decay processes experienced by trees, such as infection by fungi and wood-boring invertebrates. These processes can vary between trees in close proximity yet will greatly influence the degree to which a tree smashes upon felling. The only other study found in the literature which examined the accuracy of tree-felling surveys was done in medium height Blackbutt forest (*Eucalyptus pilularis*) in NSW, Australia (Mackowski, 1987). In that study the rates of hollow loss were considerably less than was found for the current study, although similarly larger trees showed greater error. It was estimated that 14%

of hollows were lost in trees in the 120-139 cm diameter class and 10% in trees in the 160+ cm diameter class, with little loss in other size classes (Mackowski, 1987). However, these rates of hollow loss were obtained by comparing counts pre and post felling and the error from the ground-based surveys was not taken into consideration. This means that these figures are likely to be an underestimate of the true rate of hollow loss. Yet it is expected that the rates of hollow loss in trees examined by Mackowski (1987) would be less than those of the current study as the average tree size examined was considerably smaller; approximately 40 m tall and with a maximum diameter of approximately 160 cm compared to a maximum of 70 m tall and 392 cm in diameter. This smaller size of tree was intentionally chosen by Mackowski (1987) to reduce the level of error that results from trees smashing upon being felled.

The number of branch and trunk hollows lost during post-fall surveys was significantly related to the proportion of the canopy and the trunk that could not be examined. The amount of unexamined tree was therefore used to 'correct' the number of hollows thought to have occurred in a tree. This correction procedure is obviously a simplification of real processes. Rotten or hollow wood is likely to smash more on impact in the larger trees. This may result in an underestimate for even the corrected hollow abundance figures for larger and more senescent trees. Using the amount of unexamined tree to 'correct' the number of hollows appears to be appropriate because the best model predicting the difference between pre-fall hollow counts and corrected hollow counts included the same variables as the model predicting the number of Lost hollows. The main difference between the two models was that the coefficient for tree diameter was smaller in the 'difference' model. The result of this smaller value is that the corrected hollow counts (an estimate of the true number of hollows) are likely to be an underestimate, particularly in larger trees. For similar future studies it is recommended that an accurate method for counting hollows such as tree climbing should be used to calibrate the error as per Harper *et al.* (2004). However, when this is not feasible, estimating the amount of tree unsurveyed is a potential alternative for increasing the utility and accuracy of tree-felling survey estimates.

Ground-based surveys

Ground-based, or pre-fall surveys are frequently used by managers to assess the hollow-bearing status of trees, often for the selection of suitable habitat trees for fauna. However, the results of the present study indicate that there can be substantial error in correctly identifying hollows in standing trees due to misclassification and non-detection of hollows.

The result of the present study was that 14.7% of hollow were misclassified during ground-based surveys. Hollows were mostly misclassified because they were shallower than the

minimum two centimetres required to meet the definition of being a True hollow. There is little that can be done about this under normal circumstances and it is a well acknowledged limitation of the survey method (Gibbons and Lindenmayer, 2002; Whitford, 2002). To cater for this weakness, some authors have mounted a camera on a pole to examine the hollows more closely (Soderquist and Lee, 1994). However, this technique is expensive and the equipment is unwieldy and limited to use in shorter trees. Another source of error was falsely identifying marks on the tree or indents in the bark as hollows. This error may be limited by surveying during optimal conditions (i.e. sunny days) and using stronger binoculars. The final reason hollows were misidentified was because the entrance width was smaller than the specified two centimetres. Practice at viewing hollows from a distance combined with calibration will most likely reduce this source of error.

Although minimising the rate of hollow misidentification is important it is unlikely that this source of error will be completely eliminated. Given this, a model predicting the occurrence of a misidentified hollow would be extremely useful. The variables most closely related to the likelihood of a tree containing a False hollow were the number of hollows seen in the ground-based survey and dead branch index. The more hollows that were seen during the ground-based survey the more likely it was that one was misidentified. However, dead branch index is likely to have a stronger influence as indicated by its inclusion in both the tree and hollow level models. Greater dead branch index means that more dead wood is found on the tree. More dead wood indicates greater stress is being or has been experienced by the tree. It is hypothesised that the trees with less dead wood in the crown are more likely to have shallow hollows which leads to greater misidentification. Yet despite the production of these models they cannot be used to predict the occurrence of misclassification because they explained too little variability in the data (<15%) to be useful.

Several papers have referred to the occurrence of misclassification (Whitford, 2002), but only one reference was found which estimated the rate of misclassification. In dormant oak forest in the USA, two of the 57 cavities identified (3.5%) were found to not actually be true hollows (Healy *et al.*, 1989). This result is lower than was found for the present study. However, no clear definition of a hollow was given by Healy *et al.* (1989) and it appears that no minimum dimensions were required. It is likely that this alone explains much of the difference in error rates between the two studies.

The rate of hollow non-detection was considerably higher than the rate of misclassification, with 53% of hollows being unobserved during ground-based surveys in the current study. This rate of error lies within the bounds of the results found in other studies. In *E. pilularis* forest in New South Wales, Australia, it was found that the rate of error varied with tree

diameter (Mackowski, 1987). Branch hollows began to be missed when trees reached 100 cm in diameter, with an average of 38-65% being missed depending on the size class. Few trunk hollows were missed on trees smaller than 140 cm in diameter. An average of 25% of hollows were missed for trees 140-159.9 cm in diameter and 93% were missed for larger trees. Hollows in *E. pilularis* were detected more accurately on smaller than larger trees (although this may be related to relative hollow abundance). In *E. leucoxylon* woodland in Melbourne, Australia, approximately 45-90% of hollows were not detected during ground-based surveys, the exact proportion varying with observer (Harper *et al.*, 2004). In oak stands in central Massachusetts, USA, 20% of hollow-bearing trees and 23% of hollows were not identified during ground-based surveys in the winter period (Healy *et al.*, 1989). While in mixed-species forest in Poland, only 6% of cavities were not visible from the ground (Ruczynski and Bogdanowicz, 2005).

In the current study it was expected that the accuracy of detecting hollows would vary between sites according to the density of the under and overstorey. The models indicated that the number of undetected hollows found on a tree did not vary between sites, but whether a particular hollow was detected did. The reason for this apparent inconsistency as to the effect of site is uncertain. It is possible that for the tree-level model no variation between sites was detected due to the random and differing location of hollows on trees. The variation between sites may have become apparent in the hollow-level model because the site attributes had a greater influence on those hollows that were more difficult to detect; small hollows, those located in branches high above the ground and those facing upwards. It was also found that whether a particular hollow was detected varied between trees, although this effect became insignificant upon the removal of four outlying trees. The variation between trees is possibly due to the variation in tree architecture. Dense canopies and major branching will obstruct the view from the ground, thereby influencing the accuracy of hollow identification for some trees.

The accuracy of ground-based surveys can be improved in deciduous trees by examining trees during the dormant period when the branches are bare (Healy *et al.*, 1989; Kahler and Anderson, 2006). However, this is not always practical and does not help with examination of evergreen trees. One of the most effective ways to minimise the error of non-detection is to use experienced surveyors and to increase the time taken when examining a tree, although again this will not completely eliminate such error (Harper *et al.*, 2004). Some studies have extrapolated counts of hollows made during ground-based surveys according to the amount of tree that was unobserved (Walter and Maguire, 2005). A similar approach could have been taken in the present study using the variable 'tree visibility', which is a categorical assessment of the amount of tree that was examined. However, tree visibility was not found

to be a good predictor of the number of hollows unobserved in the current study and so the validity of such a technique should be assessed if used.

The poor predictive ability of both the misclassification and the non-detection models in the current study means that no clear way of correcting the number of hollows observed during ground-based surveys was found. The most erroneous hollows were those with a small entrance diameter, particularly when they were located in the canopy of the tree. Small hollows are more numerous on trees than larger ones (Whitford, 2002) meaning the error resulting from this source may occur relatively frequently. This suggests that it may be inappropriate to use ground-based surveys as an estimate of true hollow abundance unless some sort of calibration is done. It has been found in other studies that double-sampling using an additional method such as tree-climbing can greatly improve the accuracy of hollow counts from ground-based surveys (Harper *et al.*, 2004). However, while a constant calibration rate was found in the trees examined by Harper *et al.* (2004), a calibration for the trees examined in this study is unlikely to be straightforward because tree size and senescence have been shown to influence the rates of error.

The accuracy of the ground-based survey was assessed by comparing the hollow counts to the corrected counts of hollows seen in the tree-felling survey. The correlation value found in this study (0.77 corrected, 0.65 uncorrected) lies within the range found in other papers (r = 0.57: Whitford, 2002; unspecified correlation value = 0.70 - 0.83 with an outlier of 0.57: Harper *et al.*, 2004). However, the conclusions other studies have reached on the utility of ground-based surveys vary from saying they are of little practical use (Whitford, 2002) to stating they can be an accurate and useful technique when double-sampling is done (Harper *et al.*, 2004). While it is known that the accuracy of surveys will vary with observer and the time taken to conduct the survey (Harper *et al.*, 2004), it is likely that the reason for such a large difference is related to the type of forest and trees examined. Stronger correlation was achieved in open eucalypt woodland where the trees were less than 20 metres tall and the largest diameter was about 75 cm (Harper *et al.*, 2004). Weaker correlation was found in a study examining a number of different forests including low, intermediate and high rainfall zones where the trees were larger, (up to 180 cm in diameter) and taller (up to 47 m in height) (Whitford, 2002). Hollows are less likely to be detected when located high up in the crown (Mackowski, 1987), so it is likely that the architecture of the trees and the structure of the surrounding forest contributed to the difference in ground-based survey accuracy. The trees examined in the present study were generally larger that those found in previous similar studies. In addition, the present study was done in both dry open forest and dense wet forest. Therefore the potential exists to have greater error in the current study than other similar studies. Yet despite this, the correlation coefficients achieved in this study indicate that there

was a strong relationship between the number of hollows seen in the ground-based survey and the number that are likely to exist in a tree. No indication was found that the accuracy with which hollows are detected varies between the different forest types. This indicates that ground-based surveys can still be useful, but as a measure of relative rather than actual hollow abundance.

The use of a particular tree by animals has often been related to the abundance of hollows found in the tree (Chapter 6; Lindenmayer *et al.*, 1996a). Consequently, hollow-bearing trees to be retained for habitat in production landscapes are often selected according to the presence of visible hollows (Wayne *et al.*, 2006). As ground-based surveys provide a good measure of relative hollow abundance, they can be a useful technique for managing faunal populations despite their imprecision.

Conclusions

Hollow identification is essential for ensuring that suitable habitat for Australian fauna is retained in production forests. Ground-based surveys and tree-felling surveys are two ways in which they can be studied, but the errors associated with both methods need to be taken into account when interpreting results. The results of the present study support the view that ground-based surveys should be used only as an indicator of relative hollow abundance rather than a measure of actual abundance. The more hollows seen in the ground-based survey, the more likely at least one will be misidentified and the more hollows are likely to have been undetected. The most erroneous hollows are those with a small entrance diameter, particularly when they are located in the canopy of the tree. Tree-felling surveys on the other hand can provide a more accurate estimate of hollow presence/absence and abundance, particularly when a simple correction factor is applied based on the proportion of the trunk and canopy that is not surveyed. Both survey methods can be justifiably useful for conducting studies on tree hollows but the error associated with them should always be taken into account when interpreting the results.

Chapter 5

Does hollow occurrence vary with forest type?

Abstract

Both the distribution of hollow-using fauna and the management of production forests can vary with forest type. If hollow occurrence varies with forest type, the use of generic prescriptions for managing the hollow resource may be inappropriate. This study examined the relationship between the occurrence and abundance of tree hollows and various site and tree attributes in wet and dry *Eucalyptus obliqua* forest in Tasmania using Classification Trees, Random Forests and Generalised Linear Models. The relative importance of the main hollow-forming processes for the different forest types was also considered. A total of 388 trees at 39 sites were examined before and after being felled. Evidence of invertebrate damage contributing to hollow formation was more common in wet forest than either dry or damp forest. Evidence of hollow formation by fire and fungi was more prevalent in drier than wetter forest. There was no difference between forest types in the proportion of hollows showing evidence of limb breakage. Of the site and tree factors that were practical for use in the field, the number of hollows observed before the tree was felled, tree diameter and the amount of dead wood in the canopy were the main factors associated with both hollow presence and abundance. Tree age, an important variable to consider when planning harvest rotations, was strongly associated with hollow presence but had less bearing on hollow abundance. The size of hollows that were found in a tree was related to greater senescence, as indicated by greater values of the afore-mentioned factors. Although significantly more hollows of all sizes were found in wet forest than either dry or damp forest, the age at which trees began to produce hollows was similar between the different forest types. Trees grow slightly more slowly in dry forest than wet which means that there was a slight increase in the size at which trees produced hollows in wet forest. However, this difference was not great enough to be identified in the predictive models developed. Despite this, it was concluded that for the forests examined in the present study, trees of a similar age should be retained but the retained trees should be slightly larger in wet forest than dry.

Introduction

Many animal species use tree hollows or cavities as nesting or roosting sites (Smith and Lindenmayer, 1988; Bai *et al.*, 2003; Gerhardt, 2004; Kalcounis-Ruppell *et al.*, 2005; Walker *et al.*, 2005) but there is concern about the future availability of the hollow resource, particularly in areas of production forestry (Gibbons and Lindenmayer, 1996; Ball *et al.*,

1999; Fan *et al.*, 2004). One of the concerns with production forestry is that it reduces the age of the stand and, therefore, the availability of hollows (Lindenmayer *et al.*, 1990a; Gibbons and Lindenmayer, 1996). In attempts to cater for the requirements of fauna in harvested areas, codes of practice often require that trees likely to provide faunal habitat (habitat trees) are retained (Keisker, 2000; Wayne *et al.*, 2006). Habitat trees (also known as 'wildlife trees') are generally required to contain natural cavities or have attributes indicating the future presence of cavities (Keisker, 2000; Wayne *et al.*, 2006). This is particularly important in hardwood forests and in areas lacking primary excavators of hollows (e.g. woodpeckers).

A number of studies have attempted to develop criteria to identify suitable habitat trees (Lindenmayer *et al.*, 1990c; Bunnell *et al.*, 2002; Giese and Cuthbert, 2005). This has largely been done by relating hollow occurrence and/or abundance to environmental and tree attributes. A large number of attributes have been found to be important: tree species, tree height, tree diameter, amount of dead wood in the canopy, crown size and senescence, aspect on the tree, disturbance level of the site, tree age, tree shape, tree dominance, tree lean, basal area of the site, site slope, stand age, latitude, topography, mean annual temperature, annual rainfall and fire damage (Lloyd and Lau, 1986; Mackowski, 1987; Rose, 1993; Lindenmayer *et al.*, 1993b; Bennett *et al.*, 1994; Wormington and Lamb, 1999; Gibbons *et al.*, 2000b; Fox *et al.*, 2001; Whitford, 2002; Whitford and Williams, 2002; Fan *et al.*, 2003; Wormington *et al.*, 2003; Remm *et al.*, 2006; Munks *et al.*, in press). Most of these studies indicate that all measures of tree age and senescence are positively related to the formation of natural hollows.

A number of the above-mentioned variables are related to the classification of forests into different communities or forest types. Forests with the same dominant tree species are generally classified into different communities based on the understorey vegetation. In Australia, wet sclerophyll forests are characterised by having broad-leaved species, ferns or a rainforest element dominant in the understorey, while dry sclerophyll forests have narrow-leaved, grassy or mixed understorey dominance (Ashton, 1981; Kirkpatrick *et al.*, 1988; Forest Practices Authority, 2005). The floristic composition of an area depends largely on the rainfall, soil type, aspect, topography and fire regimes (Ashton, 1981). In areas of low rainfall and poor soils, plants tend to be more scleromorphic and flammable (Ashton, 1981). More scleromorphic and flammable plants results in an increased frequency of burning which perpetuates dry sclerophyll forest. In wetter areas and those with richer soils, the plants are less sclerophyllous and so are less frequently burnt (Ashton, 1981). However, such forests accumulate litter rapidly meaning that when a fire does occur it can result in partial or complete stand replacement. Therefore, different forest communities will vary in fire

regimes, soil nutrition, rainfall and tree size and age. Given that these attributes have also been related to hollow occurrence, it is likely that the rate of hollow formation and hollow abundance will also vary between forest types.

The forest type attributes that are most likely to influence hollow formation are disturbance frequency and severity (Inions *et al.*, 1989; Adkins, 2006), exposure to wind (Harper *et al.*, 2005) and site productivity (Gibbons and Lindenmayer, 2002, p. 49). Wet forests typically experience fewer, but more intense disturbance events, such as fire. Fires assist hollow formation by creating wounds which allow the entrance of decay causing organisms, affecting the defence system of the tree and therefore its ability to heal and by enlarging existing hollows (Inions *et al.*, 1989; Adkins, 2006). It has been proposed that fire creates large pulses in the number of hollows in wet forest, but hollow loss in such areas is also expected to be high due to higher rates of decay (Gibbons and Lindenmayer, 2002, pp. 119-120). Wind damage may also contribute to hollow formation by increasing the amount of branch shedding (Harper *et al.*, 2005). Trees in dense tall forest are more exposed to wind damage at the top of the crown while open woodland trees have the entire crown exposed. Ambrose (1982) suggested that wind may be a source of difference between forest types in hollow location in younger trees. The influence of soil nutrition on hollow formation is less clear than the previous two factors. Soil nutrition is related to growth rates. It has been proposed that the rate of hollow formation is related to the growth rate of trees, with faster growing trees forming large hollows at a younger age (Wormington and Lamb, 1999). Areas of high productivity are also believed to support greater hollow densities because they have more trees per unit area and the trees tend to be larger with bigger crowns, attributes associated with hollow occurrence (Gibbons and Lindenmayer, 2002).

Exploring the relationship between forest type and hollow occurrence is of interest for two main reasons. The first is that animal density can vary between forest types (Kavanagh, 1984; Alexander *et al.*, 2002) and with stand elements related to forest type such as understorey composition (Davey, 1984; Lindenmayer *et al.*, 1990c), vegetation density (Lindenmayer *et al.*, 1996a), stand structure (Pausas *et al.*, 1995; Wayne, 2005) and tree spacing (Lindenmayer *et al.*, 1990c). The species composition and density of hollow-using fauna found in a particular forest type need to be considered when designing management prescriptions in order to ensure they are meeting the demand for the hollow resource. The second reason is that if a difference exists in the rate of hollow formation between different forest types, management prescriptions will need to reflect the types of trees that need to be retained. For example, in Victoria the number and size of trees to be retained varies between areas and forest types (Wayne *et al.*, 2006). In Tasmania no differentiation is currently made between forest types, although a difference in tree retention prescriptions occurs between the

harvesting systems, which generally corresponds with forest type (Forest Practices Board, 2000). This difference is a result of logistical considerations instead of ecological considerations. In areas with no or low intensity burning (usually dry forest areas), habitat clumps containing 2-3 habitat trees are retained every five hectares. In areas with high intensity burning (generally wet forest areas), patches of habitat 50 × 20 m in size are retained every 200 m along the boundary of the coupe. An understanding of the relationship between hollow occurrence and forest type means more specific prescriptions can be constructed, if required, to cater for the availability and demand for hollows in the different forest areas.

The relationship between broad forest type and hollow occurrence is unresolved. Studies which have considered the issue generally had little or no overlap in the dominant tree species found in the different communities (Bennett *et al.*, 1994; Wormington and Lamb, 1999; Lindenmayer *et al.*, 2000). Yet despite this, some generalisations between wet and dry forests have been made. Gibbons and Lindenmayer (2002) stated that "the number of hollows in wet sclerophyll forests will vary more widely, but on a longer cycle than dry sclerophyll forest" due to the difference in fire disturbance events found between such forests. Fox *et al.* (2001) concluded that trees growing in higher rainfall locations with more productive soils generally produce smaller hollows. Wormington and Lamb (1999) proposed that longer time periods are required to produce hollows suitable for fauna in dry forest than wet forest. Ambrose (1982) stated that the strong apical dominance of trees in tall forests results in fewer and smaller side branches than trees in more open forest, which limits the process of hollow formation. He also proposed that the location of hollows in young trees was related to the tree and forest structure, with trees in dense stands being more exposed to wind damage at the top of the crown while trees in more open forest have the whole canopy exposed to wind. These studies suggest that forest type may affect hollow occurrence, but due to the variety of effects proposed and differences in the dominant tree species found in the forest types examined it is apparent that the issue needs some clarification.

The aim of the present study was to examine the relationship between forest type, site and tree characteristics and the occurrence and abundance of tree hollows. In particular we aimed to provide information on the characteristics of sites and trees most likely to provide hollows in both wet and dry *Eucalyptus obliqua* forest in Tasmania. *Eucalyptus obliqua* has been shown to be an important tree species for hollow dependent fauna in Tasmania (Haseler and Taylor, 1993). Models developed from the results of the present work can be used to inform the revision of current guidelines (Forest Practices Board, 2000) for the maintenance of the hollow resource in *Eucalyptus obliqua* forest subject to production forestry activities. The

focus of the current chapter is at the tree level. The issue of stand level management will be addressed in the following chapters.

Methods

Study sites and forest types

Thirty-nine study sites were selected in State Forest forestry coupes that contained either wet (12 sites) or dry (27 sites) *Eucalyptus obliqua* forest (Figure 5.1). *Eucalyptus obliqua* forest was chosen for the present study because it is widespread throughout the state and both wet and dry *E. obliqua* forest are amongst the main forest types used by the forest industry in Tasmania. These broad forest types, based on vegetation mapping produced during the Tasmanian Regional Forest Agreement (Tasmanian Public Land Use Commission, 1996; Harris and Kitchener, 2005), are two of the most widespread forest types in Tasmania of use to the forest industry. In this paper a subset of dry forest sites (8 sites) are referred to as 'damp' forest because these areas showed characteristics of both wet and dry forest (Table 5.1). Dry *E. obliqua* forest is characterised as being dominated by *E. obliqua* with an understorey of sedges, heaths or narrow-leaved shrubs. Damp forest is also dominated by *E. obliqua* but with an understorey dominated by an equal mixture of broad-leaved and narrow-leaved shrubs. Wet *E. obliqua* forest has a dominant *E. obliqua* cover of more than 5%, with an understorey containing broad-leaved shrubs, rainforest species, or tall tea-trees. The 39 coupes examined were selected in consultation with forest managers and harvested between January 2004 and May 2005. Forest type was significantly related to a number of site and tree variables (results not presented). With increasing forest wetness, there is a corresponding significant increase in basal area, soil nitrogen, soil phosphorus and time since logging and a decrease in latitude, soil pH, burn damage and time since fire. The trees found in wetter forest also tended to be greater in diameter and height, older and have more dead wood than trees in drier forest.

Information on disturbance history for each site was obtained by examining records from Forestry Tasmania, questioning local foresters and on a small number of occasions, contacting nearby landowners. Of the 19 dry forest sites, no records of previous logging could be found for three (although cut stumps were found at one) and four were reported to have never been logged (although cut stumps were found at three). The time since logging for the remaining 12 sites varied between 9 and 100 years (median 35 years). Indication of the fire history was obtained for all but five sites, with time since fire ranging from 5 to 78 years (median 24). Of the eight damp forest sites, no information on previous logging could be obtained for two sites (although cut stumps were found at one), one site was reported to have never been logged and the time since logging for the remaining varied from 45 to 100

years (median 65). No fires were recorded for four of these sites, the remaining having fire histories from between 18 and 78 years (median 29). For the twelve wet forest sites, no history of logging was reported in eight (although cut stumps were found in one of these), with the time since logging ranging from 35-65 years for the remainder (median 57.5 years). No fires were known to have occurred in six of the sites, the most recent fire for the remainder ranging between 25 and 80 years (median 69.5).

Figure 5.1 Location of study sites in Tasmania, Australia, indicating forest type. Squares indicate dry forest, triangles indicate damp forest and circles indicate wet forest.

Data collection

Approximately thirteen trees with a diameter at breast height over bark (dbh) of 50 cm or greater were selected at each site. The initial tree was haphazardly selected and subsequent trees were selected as being the one that was closest to the previous but more than 20 metres away from any other tree examined. Each tree was surveyed for potential hollows from the ground using binoculars, with attempts made to view the tree on all sides. All observations were made by the author to prevent observer bias (Harper *et al.*, 2004). A hollow was defined as a cavity in the tree with a minimum entrance diameter and depth of 2 cm, located at least two metres above the ground. These dimensions were chosen because 2 cm was considered to be the minimum entrance width an animal could use, birds were observed nesting in shallow hollows (scars) approximately 2 cm deep and 2 m above the ground was considered sufficient to be able to differentiate between arboreal hollow-users and other animals. The trees were then felled by forest contractors as part of normal forestry operations. Not all trees surveyed pre-felling could be studied again post-felling due to logistical constraints (such as the trees not being felled). A total of 388 trees were successfully surveyed post-felling, between six and twelve trees at a site (average 10 trees ±

1.3 s.d.) with an average diameter of 106.7 cm (± 50.2 s.d.). Each of these trees was again searched for hollows. The dimensions of the hollows were measured whenever possible, allowing classification of hollows as small, medium or large (Table 5.2).

Table 5.1 Site and tree attributes.

Variable name	Definition
Forest type	Forest type was determined using the RFA classification from the Forest Botany Manual (Forest Practices Authority, 2005) and then reduced to the classes: dry (DRY hOB); damp (DRY shOB); wet (WET all types).
Basal area	The basal area of each stem with a DBH > 10 cm was measured in a 0.25 ha circular plot. (The actual radius used was dependent on the slope of the site to ensure a horizontal area of 0.25 ha). The results were combined and extrapolated to produce an estimate of the basal area per hectare at each site.
Latitude	A GPS was used to determine the latitude of the sites in GDA. Confirmation was obtained from a 1:25000 map of the area.
Longitude	A GPS was used to determine the longitude of the sites in GDA. Confirmation was obtained from a 1:25000 map of the area.
Altitude	A GPS was used to determine the altitude of the sites (m). Confirmation was obtained from a 1:25000 map of the area.
Aspect	A compass was used to determine the direction directly down-slope. The aspects were divided into four categories (N, S, E, W).
Slope	A clinometer was used to determine the approximate average slope of the site in degrees for the area in which the trees examined were located.
Rock	Parent rock type of the substrate was obtained from the plans developed for the harvesting of the coupe (Forest Practices Plans) and was classified as: granite; dolerite; sediment.
Soil	Soil type was obtained from Forest Practices Plans and classified as: loamy; clayey; sandy.
pH	Four soil samples were collected from the top 10 cm. The samples were mixed and the pH was measured using a probe in a 1:5 soil: distilled water solution.
Conductivity	Soil conductivity was measured as for soil pH but using a conductivity meter.
Soil nitrogen	Soil was air-dried, ground, sieved and measured for total nitrogen using the Kjeldahl method (Jackson, 1964).
Soil phosphorus	As for soil nitrogen, but measured for available phosphorus by acid-fluoride extraction (Jackson, 1964).
Topography	The average topography of the site was categorized as: ridge; upper slope; mid-slope; or lower slope / gully.
Stand age	The age of the stand was categorized according to the information from the photographic interpretation (PI) of the vegetation age structure within each plot (obtained from Forestry Tasmania's concise-PI type maps: Stone, 1998) into: mature forest; mature with regrowth; regrowth with mature; regrowth.

Table 5.1 continued

Variable name	Definition
Species	Tree species was determined.
Dbh	Tree diameter at breast height over bark (1.3 m) was measured using a diameter tape (cm).
Age	Tree age (years) was determined by ring counting when a wood sample was available. (Growth rates were extrapolated to estimate the age of the missing section if it occurred). For trees with no wood sample a growth model was used. Details are provided in Chapter 3.
Shape	A classification of the shape of a tree, based on definitions by Smith and Lindenmayer (1988): (3) tree with full round crown; (4) mature tree with major branching; (5) mature tree with dead branches; (6) mature tree with dead top but big crown; (7) mature tree with dead top and small crown; (8) burnt tree with only epicormic growth.
Crown class	The dominance of each tree was categorized according to the relative height of the trees as: dominant; co-dominant; sub-dominant / suppressed.
Burn damage	The burn damage of the tree was assessed as: (1) no evidence of fire; (2) bark damage; (3) wood exposed and damaged by fire; (4) severe fire damage where the base of the tree forms a bridge.
Dead branch index	The number of dead branches in each of the following size categories was estimated: (S) small (5-15 cm); (M) medium (15-40 cm); (L) large (> 40 cm). An estimate of dead branch index was obtained by the following formula: DBI = S + 5M + 10L.
Bark width	The maximum width of the bark (cm) was measured post-felling to accuracy of 0.5 cm using a measuring tape.
Tree height	Tree height was determined using a measuring tape and a clinometer (m). Measurements were taken about 20 m from the base of the tree. Trees around 30 m in height were most accurately measured in this way (as compared with felled trees), with taller trees being overestimated and shorter trees being underestimated.
Crown volume	The volume of the crown (m^3) as estimated using the diameter of the crown north-south and east-west and the estimated depth of the crown.
Pre-fall count	The number of potential hollows observed during the ground-based (pre-fall) survey.

Table 5.2 Hollow size category definitions. If a hollow did not meet the minimum depth for a hollow of particular entrance diameter it was placed it the next smallest hollow size category.

Hollow size	Minimum entrance diameter (cm)	Minimum depth (cm)
Hollow	2.0	2.0
Small hollow	2.0 – 5.0	5.0
Medium hollow	5.1 – 10.0	10.0
Large hollow	> 10.0	15.0

Information on the characteristics of the sites and trees were collected during the surveys (Table 5.1). These variables were chosen for their anticipated value as predictors of any differences in hollow counts. Any observed cause of hollow formation was noted. The alternatives considered were: formed by breakage of a limb on the tree, fire damage, invertebrate wood borers and fungi. Breakage of a limb was determined by the shape of the hollow and the presence of wood fragments. Fire damage was determined by the presence of charcoal. Invertebrate wood borers were determined by the presence of grooves formed by boring or a honeycomb appearance inside the hollow. The presence of fungi was determined by assessing the dampness of the hollow.

Data analysis

Forest type and hollow formation

To examine if there was a significant difference between forest types in the number of hollows found in a tree, ANOVA was done in SPSS (2005). The relationship between hollow occurrence and tree diameter and age was examined graphically. The relative importance of fire, fungi, invertebrates and limb breakage for hollow formation was examined by expressing the number of hollows believed to be formed by a particular process as a proportion of the hollows for which an assessment of formative process was made. To examine if there was a difference between the relative importance of the formative processes and forest type, Chi-square analysis was used in the statistical program SPSS.

Hollow presence

To determine which variables best predicted whether a tree contained a hollow or not, Classification Trees were created using the 'mvpart' package (Therneau *et al.*, 2006) in the statistical program R (R Development Core Team, 2006). Classification Trees were selected over other techniques for a number of reasons. They provide output that is easily interpretable and facilitates the construction of simple management guidelines. They can also readily deal with missing data, a mixture of categorical and continuous predictor variables and interactions between variables. They are robust to outliers in data, insensitive to the distribution of the input data and are useful both for data exploration and for prediction (De'Ath and Fabricius, 2000; Hastie *et al.*, 2003).

Classification Trees were constructed for the presence or absence of total hollows, small hollows, medium hollows and large hollows. To select the best Classification Tree, a maximal tree was constructed (a dendrogram with the maximum number of splits possible under the conditions imposed). The tree was then cross-validated by site (4-5 sites were

combined to make 9-fold cross-validation). The optimal tree size was selected by examining the change in the cross-validation relative error in terms of tree size (De'Ath and Fabricius, 2000). The data were then fitted using the restricted tree size and the rates of misclassification were determined. The alternate primary and surrogate splits were then examined. The final model was selected as that with the lowest misclassification rate of those with sensible ecological pathways. Two sets of models were constructed. The 'causal' models used all variables except pre-fall hollow counts. The intention was to produce a model that explored the variables which may have contributed to the formation of a hollow. The 'survey' models included pre-fall hollow counts and excluded tree age, site basal area, soil nitrogen, phosphorus, pH and conductivity. The intention was to produce a model using only variables that can be easily determined in the field, thereby producing a useful management tool for estimating hollow occurrence. However, surveying trees for hollows can be time consuming and will not always be a viable option for all forest managers. Consequently two different survey models were produced; The 'high cost' survey model used all the variables stated above and took a high survey effort and the 'low cost' survey model excluded the variable 'pre-fall hollow counts' and therefore took a lower survey effort.

While Classification Trees have a number of advantages over other statistical modelling techniques, they can have lower predictive ability in some cases. A related method that can have greater predictive ability is Random Forest, done using the R package 'randomForest' (Liaw and Wiener, 2002; Prasad *et al.*, 2006). Random Forest is a process where a large number of Classification Trees are grown (in this case 1,000), with a randomised subset of predictors. The results are averaged across the trees. While this method provides greater accuracy in model predictions, it is less intuitive and less easy to use. Therefore, Random Forests were not used as the final output of the models. Instead, they were fitted to explore the relative importance of the predictor variables and to examine how much predictive ability was lost by using a simple Classification Tree. The Gini index was used to indicate the relative importance of predictor variables. The Gini index is a measure of the total decrease in node impurity resulting from the splitting of the variable, averaged over all trees. Node impurity refers to the proportions of responses in each category of the dependent variable that occur at each node or 'branch' (De'Ath and Fabricius, 2000). Therefore, the larger the value of the Gini index, the greater the contribution the variable has to the accuracy of the model. In order to compare the predictive ability of the Random Forests and the Classification Trees, results are presented as the rate of misclassification for each.

Hollow abundance

The number of hollows found in the tree after felling was not an accurate count of hollow abundance (Chapter 4). Inaccuracy occurred because sections of the tree could not be examined due to smashing, or because sections were obscured by the ground or debris. The trees smashed to varying degrees meaning that no assumption of equal error could be made. In order to correct the number of hollows likely to have occurred on the tree, the proportion of both the trunk and the canopy that could not be inspected during the post-felling survey was estimated. The assumption was made that the distribution of hollows was constant across both the trunk and the canopy of the tree. The corrected number of hollows on the trunk was then estimated by dividing the number of hollows observed on the trunk by the proportion of the trunk that was successfully inspected. This process was repeated for the canopy and the results were summed to produce the total estimate of hollow abundance for the tree (see Chapter 4 for details). This correction was made for counts of total hollows, small hollows, medium hollows and large hollows.

To determine which of the predictor variables were most closely related to the number of hollows estimated to have occurred on the tree, hierarchical partitioning was done using the 'hier.part' library in R (Walsh and Mac Nally, 2005). The results indicated that the most important variables influencing hollow abundance, in order of importance, were pre-fall hollow counts, tree diameter, dead branch index, tree age and crown volume. However, all of these variables were significantly correlated with each other ($p < 0.05$), although crown volume with less strength than the others having a Pearsons' correlation value < 0.5 for all variables. Of these variables it was decided that pre-fall hollow counts, tree diameter and tree age were the most useful variables to examine and so were used to construct separate models. Tree age was selected because it is a useful attribute to examine when considering the impact of harvesting rotations on hollow availability. Tree diameter was selected because it is a standard inventory technique in production forests and so may be useful for both habitat tree selection and for constructing predictive maps (as per Munks *et al.*, in press). Pre-fall hollow counts was selected because it has been shown in other studies to be a strong and useful indicator of hollow abundance and would therefore be a practical method for selecting habitat trees (Harper *et al.*, 2004).

In order to construct models which could predict the number of total, small, medium and large hollows estimated to have occurred on a tree, generalised linear models with a quasipoisson distribution and a log link were done using the 'nlme' library in R (Pinheiro *et al.*, 2006). Prior to the construction of these models, 20 trees were randomly selected and removed from the data set to provide a smaller, independent data set with which to test the predictive ability of the models. The remaining data set was used to construct the models.

Initially, generalised additive models were constructed using the 'mgcv' library in R (Wood, 2006). Continuous variables were smoothed for their addition into the model in order to assess whether a linear fit was appropriate and whether a transformation was required. An appropriate linear fit was achieved for all variables and so the final results are presented as a generalised linear model.

To select the best model, one of the most influential variables (age, pre-fall counts or diameter) was first fitted. Then the other variables outlined in Table 5.1 and all two-way interactions which were deemed by the author to be ecologically meaningful were added in a forward step-wise manner. Exceptions were that the variable 'age' was not considered for the pre-fall count model and vice versa, and neither were considered for the dbh model. These exceptions were made to produce one 'causal' and two 'survey' models (high cost and low cost). The selection of variables to be retained in the model was done using the change in deviance explained by the model. A variable was required to increase the deviance explained by the model by at least 5% in order to be retained, where the deviance explained was divided by the degrees of freedom. The sequential fitting of variables allowed identification of the independent effects of variables, justifying the inclusion of correlated variables in the same model.

Results

Hollow formation and forest type

There was a significant difference between forest types in the number of hollows found in trees for all hollow sizes (Table 5.3). However, this difference was only significant between wet forest and the others, with wet forest having more hollows (Figure 5.2). There was an increase in the size (tree diameter) and age of trees with forest wetness (see Figures 3 and 4). Examining the proportion of trees in each age class that contain hollows indicates that the wet and dry forests have a similar rate of hollow production (Figure 5.3). There was some indication that there were more hollows in damp forest sites, although the sample sizes were also smaller. Examination of hollow occurrence in relation to tree diameter (Figure 5.4) reveals that for a particular diameter class, a higher proportion of trees in dry forest contain hollows than was found for wet forest. The youngest tree estimated to have a hollow in dry forest was 57 ± 2 years, in damp forest was 60 ± 2 years, and in wet forest was 74 ± 3 years. The youngest tree estimated to have a large hollow in dry forest was 110 ± 8 years, in damp forest was 144 ± 8 years, and in wet forest was 81 ± 3 years. The oldest tree to not have a hollow was 258 ± 18 in dry forest, 120 ± 4 in damp forest and 289 ± 22 in wet forest. The largest tree to not have a hollow in dry forest was 110 cm in diameter, in damp forest was 98 cm and in wet forest was 133 cm.

An assessment of the cause of formation was made for 70% of the 2,357 hollows examined. There was evidence of branch breakage for 54.4% of the hollows examined, evidence of invertebrate borers for 33.7% of hollows, charcoal was found in 14.0% of hollows and 11.5% of the hollows were damp (suggesting the presence of fungi). There was a significant relationship between forest type and whether a hollow was formed by fire ($\chi^2 = 49.96$, $p < 0.001$, $df = 2$), termites ($\chi^2 = 15.45$, $p < 0.001$, $df = 2$) and fungi ($\chi^2 = 11.68$, $p = 0.003$, $df = 2$). Evidence of hollow formation by fire and fungi was more prevalent in drier than wetter forest, although the effect of fungi was only very slight. Termite evidence was more common in wet forest than either dry or damp. There was no significant difference between forest types in hollows formed by limb breakage ($\chi^2 = 3.67$, $p = 0.16$, $df = 2$).

Table 5.3 The results of ANOVA analysis between hollow abundance and forest type (df = 2).

Hollow size	F Statistic	p-value
Total hollows	13.76	< 0.001
Small hollows	6.81	0.001
Medium hollows	10.41	< 0.001
Large hollows	6.39	0.002

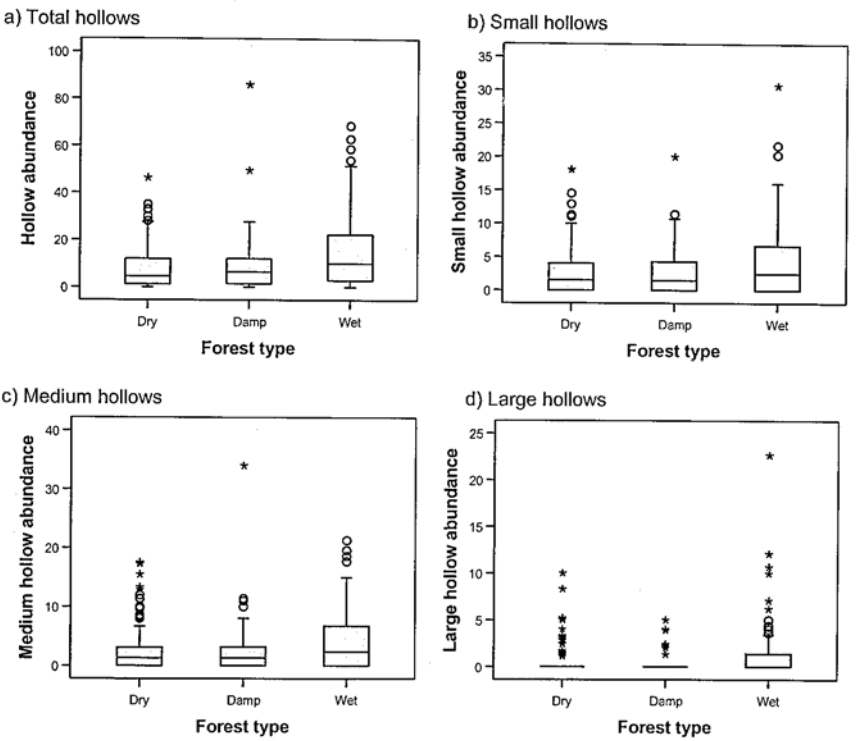
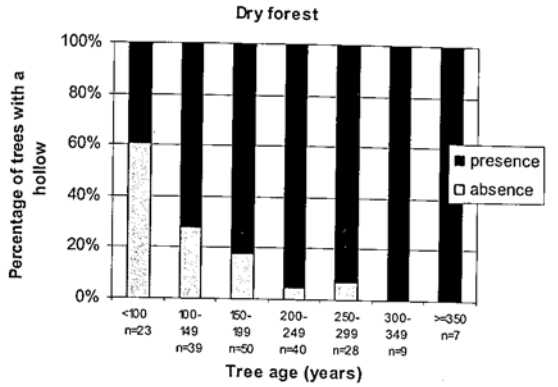
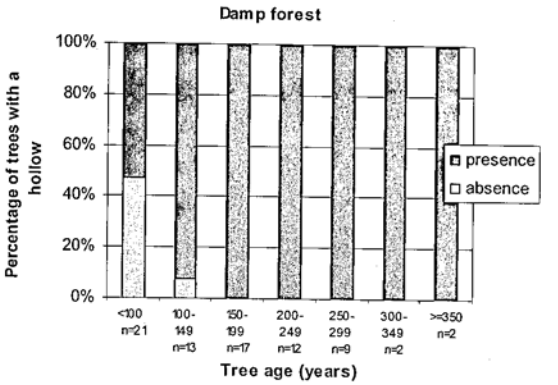


Figure 5.2 The relationship between forest type and the abundance of (a) total, (b) small, (c) medium and (d) large hollows. The box length is the interquartile range. The circles are outliers with values between 1.5 and 3 box lengths from the upper or lower edge of the box. The stars are outliers with values more than 3 box lengths from the upper or lower edge of the box.

a)



b)



c)

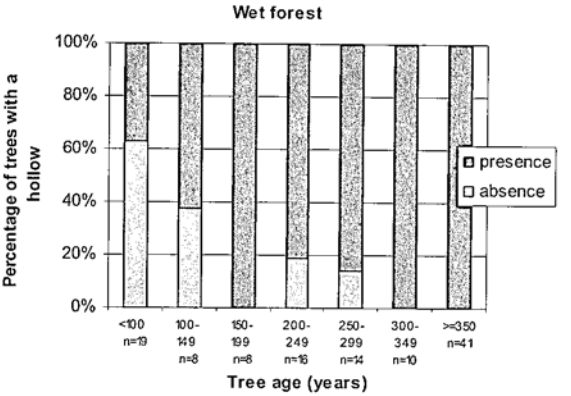
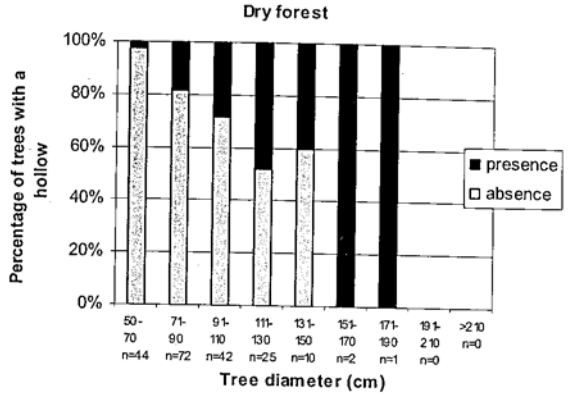
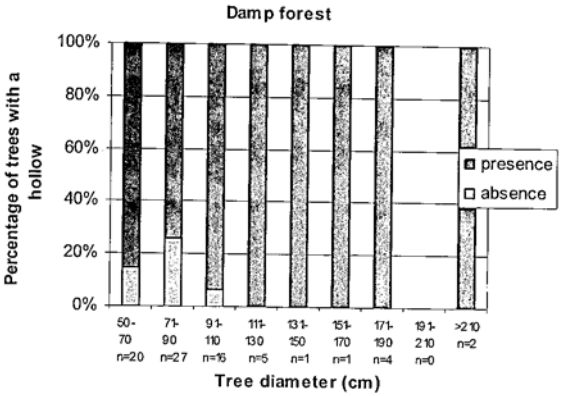


Figure 5.3 The relationship between tree age and the occurrence of hollows for (a) dry forest, (b) damp forest and (c) wet forest.

a)



b)



c)

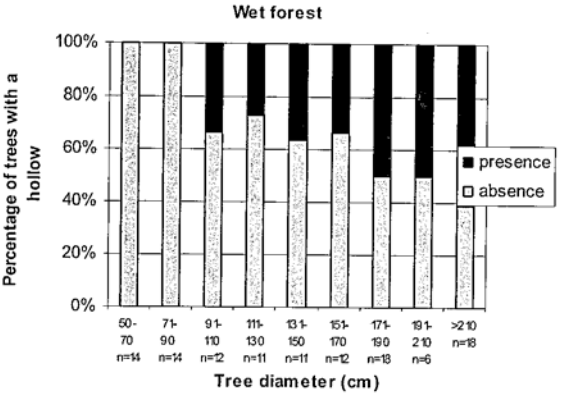


Figure 5.4 The relationship between tree diameter and the occurrence of hollows for (a) dry forest, (b) damp forest and (c) wet forest.

Hollow presence 'causal' models

For the 'causal' models, the primary split for total, small and medium hollows was tree age (Figure 5.5). This split was made at approximately 100 years for both total and small hollows, but increased to 140 years for medium hollows. For large hollows, the abundance of dead wood in the tree was a stronger influencing variable than the age of the tree. It therefore appears that trees less than 74 years are unlikely to have a hollow as it is defined in this paper. If between 74 and 100 years old, the likelihood of having a hollow is related to the size of the tree, with larger trees being more likely to have a hollow. At 100 years they are still unlikely to have a hollow that is 2-5 cm in entrance diameter and 5 cm deep (i.e. small). When older than 100 years, the chance of having a small hollow increases with tree diameter and decreases with crown volume and soil pH. Trees less than 140 years are only likely to have a medium-sized hollow (entrance diameter 5-10 cm, depth 10 cm) if there is an abundance of dead wood in the tree. For a large hollow to occur, it appears a tree must have an abundance of dead wood in the canopy and be large in diameter.

The results of the Random Forest also indicated that tree age was the most influential variable for all hollow sizes (Table 5.4). However, the relative importance of age decreased with increasing hollow size. Tree diameter and dead branch index were the next most important variables, having a similar influence to each other in all four models. Crown volume and tree height appeared to also have some importance, although largely for small and medium-sized hollows respectively. Forest type was not one of the more strongly related variables.

The misclassification rates for the Classification Trees were 11.9% for total hollows, 21.4% for small hollows, 24.5% for medium hollows and 19.1% for large hollows. An improvement on misclassification over the Null model when using the Classification Tree was greatest for small and medium hollows and least for total and large hollows. Misclassification rates of hollow absence were greater than that for hollow presence for total hollows, small hollows and medium hollows. There was greater misclassification of hollow presence than absence for large hollows (Table 5.5). The misclassification rates produced by the Random Forest were overall slightly greater than those produced by the Classification Trees; 15.3%, for total hollows, 27.2% for small hollows, 27.5% for medium hollows and 21.1% for large hollows (Table 5.5). Exceptions to this are that the Random Forest model had lower misclassification error for predicting the absence of medium and large hollows.

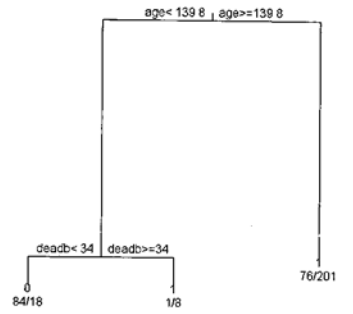
a) Total hollows

Misclass rates Null = 0 178 Model = 0 119

b) Small hollows

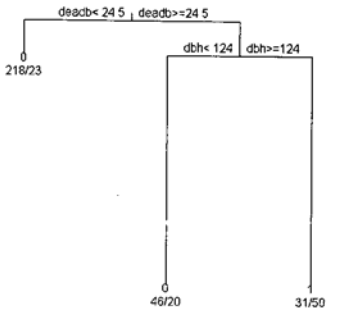
Misclass rates Null = 0 358 Model = 0 214

c) Medium hollows



Misclass rates Null = 0 415 Model = 0 245

d) Large hollows



Misclass rates Null = 0 240 Model = 0 191

Figure 5.5 The Classification Tree models using the 'causal' variables for the presence of (a) total hollows, (b) small hollows, (c) medium hollows and (d) large hollows. 'Age' indicates tree age. 'Dbh' indicates tree diameter. 'Cvol' indicates tree crown volume. 'Ph' indicates soil pH. 'Deadb' indicates the dead branch index of the tree. The vertical length of the branch represents the relative strength of that split. The zero or one at the end of each branch represents whether that leaf predominantly represents hollow absence or presence respectively. The numbers below that represent the number of samples in the current data set that were found at that leaf, where the number on the left side of the slash is the number of samples in which no hollows were found and the number on the right of the slash is the number of samples in which a hollow was found. For the Misclass rates, the Null rate is the error that occurs when it is assumed that the more prevalent result is correct in all situations. The Model rate is the misclassification rate found from using the stated model.

Table 5.4 The ten most important variables for predicting hollow presence when using the 'causal' data set, as indicated by the Random Forest for total hollows, small hollows, medium hollows and large hollows. The values presented are the total decrease in the Gini index (node impurity).

All hollows		Small hollows		Medium hollows		Large hollows	
Variable	Gini	Variable	Gini	Variable	Gini	Variable	Gini
Tree age	22.81	Tree age	23.94	Tree age	28.78	Dbh	18.35
Dbh	11.80	Dead branch	21.06	Dbh	20.95	Dead branch	17.14
Dead branch	10.87	Dbh	20.51	Dead branch	20.23	Tree age	15.91
Crown volume	7.20	Crown volume	13.78	Tree height	12.60	Tree shape	9.07
Bark width	6.75	Tree height	12.70	Crown volume	12.36	Crown volume	9.04
Tree height	6.13	Bark width	8.82	Bark width	8.66	Tree height	8.95
Tree shape	3.99	Tree shape	7.32	Tree shape	7.90	Bark width	5.83
Soil nitrogen	3.73	Soil nitrogen	5.27	Soil phosphorus	6.15	Burn damage	5.73
Longitude	3.60	Latitude	5.27	Soil nitrogen	5.98	Longitude	4.92
Soil pH	3.26	Soil pH	5.21	Burn damage	5.68	Site slope	4.89

Table 5.5 Comparing the misclassification rates when using Classification Trees and Random Forests when using the causal data set to predict the presence of total hollows, small hollows, medium hollows and large hollows.

			Classification Tree	Random Forest [a]
Total hollows		Absence	0.609	0.612
		Presence	0.013	0.048
Small hollows	True result	Absence	0.367	0.500
		Presence	0.129	0.140
Medium hollows		Absence	0.478	0.392
		Presence	0.079	0.188
Large hollows		Absence	0.105	0.074
		Presence	0.462	0.629

[a] This analysis was done excluding the variables 'basal area' and 'conductivity' because data was not available for all sites. The sample size is only 360 for the analyses because data was missing for some variables: tree height, crown volume and bark width.

Hollow presence 'survey' models

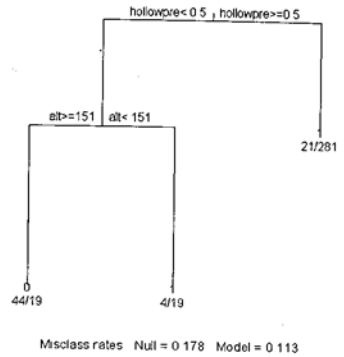
For the 'survey' models, the primary split for all models was the number of hollows seen in the pre-fall survey, although more hollows needed to be seen for increasing hollow size (Figure 5.6). For total and small hollows, if even one hollow was seen in the pre-fall survey it was highly likely that the tree did in fact have a cavity that met the definition of a hollow. If no hollow was seen, a tree was still likely to contain a hollow if found at low altitudes and contain a small hollow if the tree was around 100 cm in diameter. In order to select a tree with a high chance of containing a medium-sized hollow, more than two hollows need to be seen during the pre-fall survey. If only one or two hollows can be seen, the chances increase if the tree is very senescent (tree shape 7) or if it has a large diameter (more than 140 cm). In order to detect a tree likely to have a large hollow, the tree must be more than 120 cm in

diameter and more than six hollows need to be seen during the pre-fall survey. The misclassification rates were similar to those for the 'causal' models at 11.3% for total hollows, 23.2% for small hollows, 23.2% for medium hollows and 17.5% for large hollows (Table 5.7). For total and small hollows, trees without hollows were more frequently classified as having hollows, than trees with hollows were classified as not having hollows. The opposite was true for trees containing medium and large hollows.

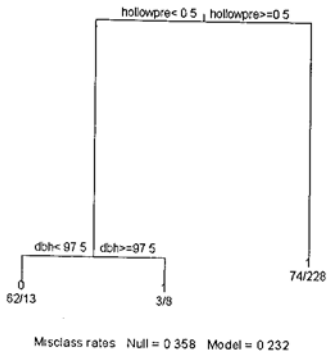
When the variable 'pre-fall hollow counts' was removed (to produce a less time costly survey model), all models except the number of medium-sized hollows indicated dead branch index as the primary branch. There was only a small difference in the improvement of the model if dead branch index was used over tree diameter and tree diameter is more practical measure for conducting field surveys, so dead branch index was also removed from the modelling. The results indicated that an increase in tree diameter was associated with an increase in hollow size (Figure 5.7). Trees larger than 80 cm in diameter were likely to have a hollow, while those smaller were likely to have a hollow if they were on north-facing slopes or they showed some burn damage. Trees larger than 86 cm in diameter were likely to have a small hollow, while those smaller had an increased chance of having a hollow if they were particularly short. Trees larger than 102 cm were likely to have a medium-sized hollow, while those that were smaller may have one if they had an abundance of dead wood in the canopy. Trees needed to be more than 120 cm in diameter to have a high probability of containing a large hollow and, even then, they were most likely to occur only at low altitudes (Figure 5.7). The misclassification rates for these models were slightly greater than those found when using estimates of pre-fall hollow counts in the survey; 14.7% for total hollows, 25.5% for small hollows, 27.8% for medium hollows and 18.6% for large hollows.

The results of the Random Forest were similar to those for the 'causal' model, except that pre-fall hollow counts replaced age in being the most influential variable (Table 5.6). When comparing the misclassification rates of the Random Forest with those for the Classification Trees the results are again similar, but slightly variable; 14.7% for total hollows, 23.6% for small hollows, 24.7% for medium hollows and 21.7% for large hollows. Exceptions to this generalisation are that the Random Forest had lower misclassification rates than both the Classification Tree 'survey' models when predicting the absence of total hollows, the presence of small hollows and the absence of medium hollows and lower misclassification rates than the tree diameter model only when predicting the absence of large hollows (Table 5.7).

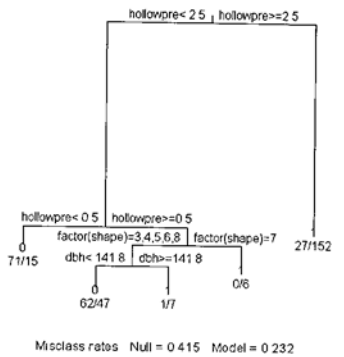
a) Total hollows



b) Small hollows



c) Medium hollows



d) Large hollows

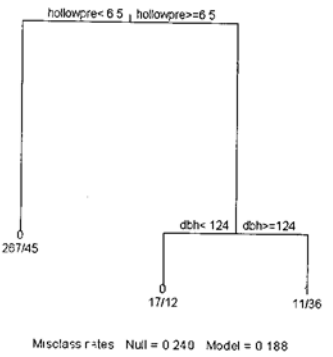
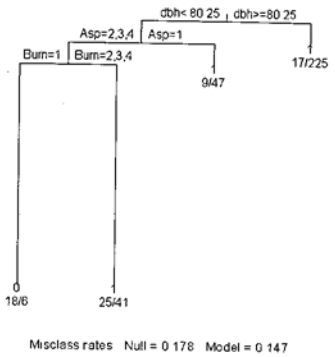


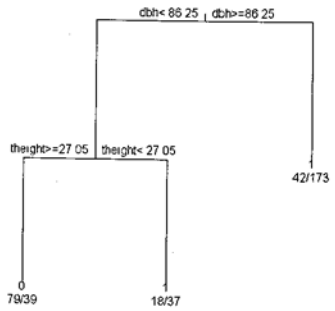
Figure 5.6 The Classification Tree models using the 'survey' variables for the presence of (a) total hollows, (b) small hollows, (c) medium hollows and (d) large hollows. 'Hollowpre' indicates the number of hollows seen in the pre-fall survey. 'Alt' indicates the altitude of the site. 'Dbh' indicates tree diameter. 'Shape' indicates the shape of the tree (Table 5.1). The vertical length of the branch represents the relative strength of that split. The zero or one at the end of each branch represents whether that leaf predominantly represents hollow absence or presence respectively. The numbers below that represent the number of samples in the current data set that were found at that leaf, where the number on the left side of the slash is the number of samples in which no hollows were found and the number on the right of the slash is the number of samples in which a hollow was found. For the Misclass rates, the Null rate is the error that occurs when it is assumed that the more prevalent result is correct in all situations. The Model rate is the misclassification rate found from using the stated model.

a) Total hollows



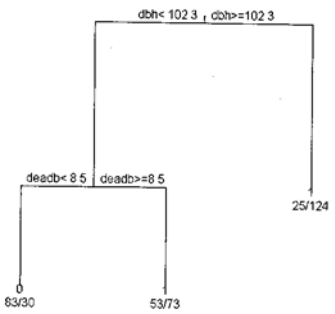
Misclass rates Null = 0 178 Model = 0 147

b) Small hollows



Misclass rates Null = 0 358 Model = 0 255

c) Medium hollows



Misclass rates Null = 0 415 Model = 0 278

d) Large hollows

Misclass rates Null = 0 240 Model = 0 186

Figure 5.7 The Classification Tree models using the 'survey' variables excluding pre-fall hollow counts for the presence of (a) total hollows, (b) small hollows, (c) medium hollows and (d) large hollows. The variables 'pre-fall hollow count' and 'dead branch index' were removed (dead branch index was not removed from the medium-sized hollow model because it did not default as the primary split) to produce a model with tree diameter as the primary split. 'Dbh' indicates tree diameter. 'Asp' indicates site aspect. 'Burn' indicates burn damage of the tree. 'Theight' indicates the height of the tree. 'Deadb' indicates dead branch index. 'Alt' indicates the altitude of the site (Table 5.1). The vertical length of the branch represents the relative strength of that split. The zero or one at the end of each branch represents whether that leaf predominantly represents hollow absence or presence respectively. The numbers below that represent the number of samples in the current data set that were found at that leaf, where the number on the left side of the slash is the number of samples in which no hollows were found and the number on the right of the slash is the number of samples in which a hollow was found. For the Misclass rates, the Null rate is the error that occurs when it is assumed that the more prevalent result is correct in all situations. The Model rate is the misclassification rate found from using the stated model.

Table 5.6 The ten most important variables for predicting hollow presence when using the 'survey' data set, as indicated by the Random Forest for total hollows, small hollows, medium hollows and large hollows. The values presented are the total decrease in the Gini index (node impurity).

All hollows		Small hollows		Medium hollows		Large hollows	
Variable	Gini	Variable	Gini	Variable	Gini	Variable	Gini
Pre-fall count	16.98	Pre-fall count	24.22	Pre-fall count	27.71	Pre-fall count	19.35
Dbh	8.89	Dead branch	16.15	Dead branch	15.97	Dbh	14.27
Dead branch	8.02	Dbh	15.06	Dbh	14.77	Dead branch	11.60
Crown volume	6.18	Crown volume	10.00	Tree height	10.21	Crown volume	6.29
Bark width	5.21	Tree height	8.23	Crown volume	9.58	Tree height	6.14
Tree height	5.13	Bark width	6.04	Bark width	6.60	Tree shape	6.07
Tree shape	3.02	Tree shape	5.79	Tree shape	5.14	Burn damage	4.85
Burn damage	2.61	Altitude	3.86	Soil nitrogen	3.90	Bark width	3.76
Basal area	2.61	Soil phosphorus	3.76	Latitude	3.44	Latitude	3.67
Latitude	2.59	Soil nitrogen	3.73	Soil pH	3.38	Site slope	3.32

Table 5.7 Comparing the misclassification rates of Classification Trees and Random Forests when using the 'survey' data sets to predict the presence of total hollows, small hollows, medium hollows and large hollows.

			Classification Tree (pre-fall counts)	Classification Tree (tree diameter)	Random Forest [a]
Total hollows		Absence	0.597	0.739	0.362
		Presence	0.044	0.019	0.060
Small hollows	True result	Absence	0.470	0.432	0.554
		Presence	0.101	0.157	0.052
Medium hollows		Absence	0.340	0.484	0.174
		Presence	0.179	0.132	0.273
Large hollows		Absence	0.081	0.064	0.037
		Presence	0.629	0.570	0.613

[a] The sample size is only 360 for the analyses because data was missing for the variables tree height, crown volume and bark width.

Hollow abundance models

The relationship between hollow abundance and tree age showed an initial sharp increase in hollow abundance with age, after which there was a plateau. The point of this plateau roughly corresponded to the age at which the hollow presence models above predicted that hollows would start to appear in a tree. Consequently, these ages were used to define a subset of the data for which the effect of age on hollow abundance would be examined. For the total and small hollow models the subset included all trees that were 100 years or older. For the medium and large hollow models the data subset used only trees that were at least 140 years old. However, once the subset of data was used to create the model, the effect of age became negligible and was replaced by dead branch index (Table 5.8a). (Tree diameter provided an equivalent model but results are not presented). For all of pre-fall hollow counts,

tree diameter and dead branch index, an increase in value resulted in an increase in hollow abundance, but the rate of increase declined (Figure 5.8).

The models indicated that, in general, greater senescence (as indicated by any measures of age, dead branch index, tree diameter or pre-fall hollow counts) was required for increasing hollow size. The predictive ability of the models, as determined by examining the fit for the excluded trees, indicated that the models had good predictive ability for total hollows. However, the ability of the model to identify trees with numerous hollows decreased with hollow size. The model was also especially poor for identifying large hollows. Of the three main models considered, the pre-fall hollow models (which included tree diameter for total hollows and dead branch index for large hollows) generally explained more of the deviance in the data and exhibited slightly smaller percentile values (Table 5.8c). The model with tree diameter alone was the next best. Forest type, either alone or interacting with another variable did not contribute significantly to the model.

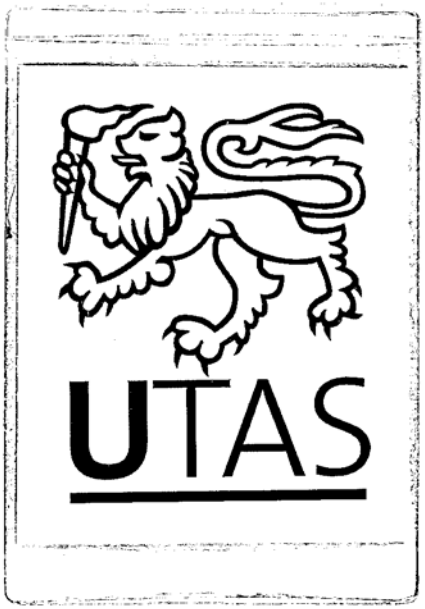


Table 5.8 The results of the quasipoisson GLM analysis on the corrected number of total, small, medium and large hollows found in a tree using (a) the log of tree age as the primary variable, (b) the log of tree diameter as the primary variable and (c) the number of hollows seen in the tree before it was felled as the primary variable. The values given are the variable coefficients and their associated standard error. [a, b]

	Variable	Total hollows	Small hollows	Medium hollows	Large hollows
a)	Intercept	0.44 ± 0.18	-0.17 ± 0.22	-0.68 ± 0.28	-3.44 ± 0.57
	Log (age)	-	-	-	-
	Log (Dead branch index +1)	0.59 ± 0.05	0.45 ± 0.06	0.57 ± 0.07	0.92 ± 0.14
	Deviance explained	36.8%	17.6%	22.5%	23.1%
	Predictive error [c]	1.88 ± 10.54	0.64 ± 3.91	1.72 ± 4.94	3.81 ± 5.18
	20 and 80% percentiles [d]	-8.35 – 9.03	-2.99 – 3.50	-1.81 – 6.07	0 – 9.55
b)	Intercept	-5.32 ± 0.51	-5.64 ± 0.61	-7.10 ± 0.69	-11.42 ± 1.22
	Log (dbh)	1.61 ± 0.10	1.44 ± 0.13	1.72 ± 0.14	2.30 ± 0.24
	Deviance explained	40.3%	26.3%	30.8%	28.4%
	Predictive error	1.67 ± 8.51	0.60 ± 3.83	1.60 ± 4.17	3.82 ± 5.01
	20 and 80% percentiles	-5.05 – 5.85	-2.23 – 3.28	-1.82 – 5.19	-0.26 – 9.27
c)	Intercept	-2.38 ± 0.44	-0.08 ± 0.14	-0.73 ± 0.16	-4.27 ± 0.48
	Log (Pre-fall count +1)	0.80 ± 0.05	0.83 ± 0.07	1.11 ± 0.07	0.81 ± 0.17
	Log (dbh)	0.72 ± 0.10	-	-	-
	Log (Dead branch index +1)	-	-	-	0.72 ± 0.14
	Deviance explained	63.5%	32.8%	44.9%	39.9%
	Predictive error	1.67 ± 6.60	0.68 ± 3.32	1.63 ± 4.40	3.77 ± 4.84
	20 and 80% percentiles	-2.32 – 6.48	-1.33 – 2.44	-1.17 – 5.47	-0.08 – 9.31

[a] '-' indicates this variable did not explain more than 5% of the deviance in the data for this model and so was not included. All variables presented had a highly significant influence ($p < 0.001$). The percentage deviance in the data that was explained by the model is provided.

[b] For model (a), n = 326 for total and small hollows because only trees estimated to be 100 years old or older were used. For medium and large hollows, n = 277 because only trees estimated to be at least 140 years old were used. For models (b) and (c), n = 388.

[c] The predictive error is the mean and standard deviation of the residual (difference between the observed and predicted number of hollows) in the removed twenty trees.

[d] The percentiles are the 20% and 80% percentiles of the predictive error.

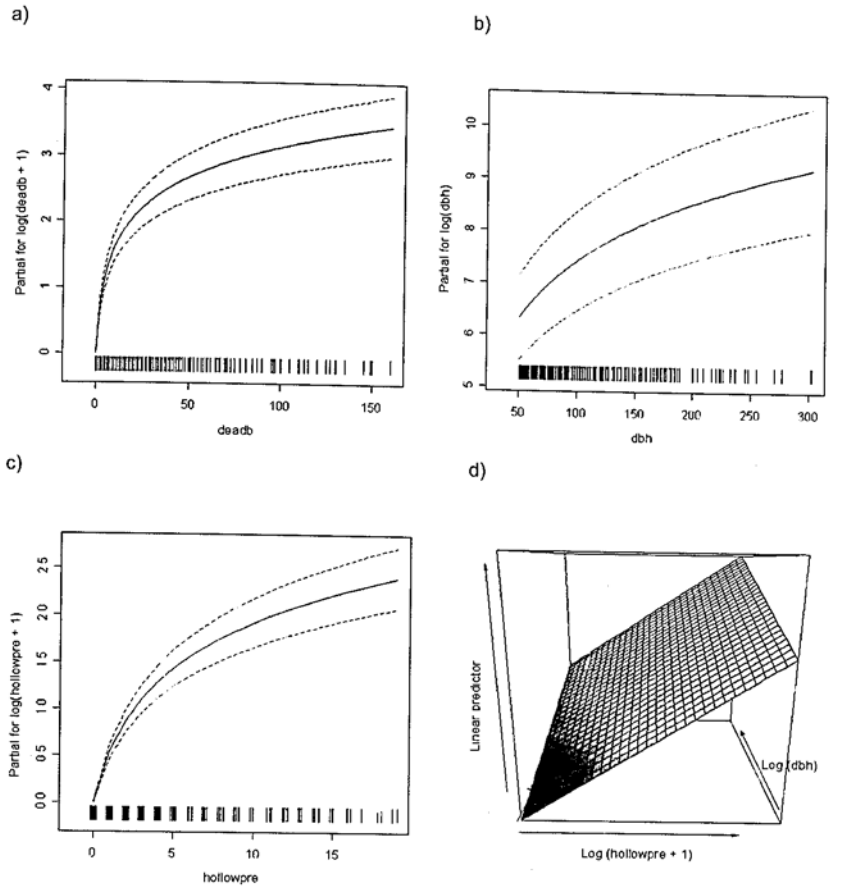


Figure 5.8 Plots of the modelled relationship between the total abundance of hollows and (a) dead branch index (deadb), (b) tree diameter (dbh), (c) pre-fall hollow counts (hollowpre) and (d) a diagram of the model including both pre-fall counts and tree diameter. The dashed lines in (a), (b) and (c) indicate approximately two standard errors of the predicted relationship.

Discussion

Significantly more hollows were found in wet forest than in either dry or damp *E. obliqua* forest in the present study. This was true for all hollow sizes. However, the rate of hollow production for trees greater than 50 cm in diameter appeared to be similar in the different forest types (Figure 5.3). The variables found to be most useful in predicting hollow occurrence across all of the broad forest types were the number of hollows seen in the standing tree, tree diameter, tree age and the amount of dead wood in the canopy. Although forest type was strongly related to all of these variables, forest type was not found to be an important variable itself in the predictive models. It therefore appears that the difference in hollow abundance between the forest types is related to differences in the size, age and senescence of the trees found in these areas as opposed to being an artefact of forest type itself.

It is slightly surprising that no differentiation was made between forest types in any of the models considered because tree growth rates vary between the forest types; trees in dry forest generally growing slightly slower than those in wetter forest (Chapter 3). This difference in growth rates was seen in the current results with trees producing hollows at a similar age but at slightly different sizes between the different forest types (Figures 5.3 and 5.4). Possible reasons for the lack of differentiation between forest types in the tree diameter model (Figure 5.7) may be due to the large amount of overlap in growth rates. This would mean that the onset of hollow production is not sufficiently different between the forest type categories to be classified separately. The results suggest that rates of hollow production are greater in damp forest than either dry or wet forest. It is unclear why rates of hollow development in such areas would be markedly different to that in the other forest types and it is possible that the results were confounded by the lower number of trees examined in damp forest.

The conclusion reached in the present study, that rates of hollow development are similar in the different forest types despite a difference in growth rates, is in contrast to that of Wormington and Lamb (1999). They reviewed a number of studies looking at development of large hollows in Australian forests and proposed that hollow development is related to growth rates, with slower growing species taking more time to develop hollows (Wormington and Lamb, 1999). In their own study comparing wet and dry forests, they found that the onset of hollow production occurred at a similar age for the three tree species considered, but that the rate of large hollow formation varied between species, but not consistently with forest type. The results of the present study indicate that differences in the diameter at which hollows are formed can occur in the different forest types, but that the rate

of formation is similar for a particular species. It is therefore proposed that differences in hollow occurrence previously found between different forest types may be a consequence of the dominant tree species examined, the disturbance history of the area and/or differences in geographic location as opposed to differences in growth rate per se (Ambrose, 1982; Bennett *et al.*, 1994; Wormington and Lamb, 1999; Munks *et al.*, in press).

Although we found similar rates of hollow formation in wet and dry forest, the processes contributing to hollow formation varied significantly between the different forest types examined. Although no differentiation could be made between primary and secondary agents of hollow formation, fire appeared to contribute more to hollow formation in dry forest than wet forest, as was expected given the known difference in fire regimes (Bowman and Kirkpatrick, 1984). There was an indication that fungi also contributed more to hollow formation in dry forest, as determined by the dampness of the hollows examined. However, this result is inconclusive because the presence of fungi can be difficult to determine (Hopkins *et al.*, 2006). Invertebrates contributed more to hollow formation in wet forest but it is unclear if this was an artefact of forest type or was really a product of tree size as the damage caused by invertebrates in Tasmania can increase with tree diameter (Elliott and Bashford, 1984). Of the formative processes considered, branch shedding was found to be the most important contributor to hollow formation as has been found in other studies (Whitford, 2002). No difference in the effect of branch shedding between forest types was detected which may explain why forest type itself did not influence hollow formation. For other studies considering the relationship between hollow occurrence and forest types, differences in hollow occurrence may be attributable to differences in the rate at which branches are shed by the dominant tree species (as was proposed by Munks *et al.*, in press).

There are many theories as to why eucalypts continually shed branches. These have been summarised in Gibbons and Lindenmayer (2002) and include competition, structural limitation and physical damage including the influence of fire and wind. As discussed, fire damage differed between forest types but appeared to contribute relatively little to hollow formation. The amount of wind damage is also believed to vary between forest types (Ambrose, 1982), but was not measured in the present study. A relationship between site aspect and hollow occurrence was related to prevailing winds by Harper *et al.* (2005) but the relationship with aspect found in the current study is more likely to be associated with fire. Small trees were more likely to have a hollow if they were on north-facing slopes and the prevailing winds that are associated with fire are north-westerly (Smith, 1998). However, aspect was only identified as important for predicting the occurrence of a hollow (many of which are obvious fire scars), irrespective of size. It was not identified as being important for the identification of larger hollows.

It therefore appears that the factor which contributed most to hollow formation was branch shedding, but the propensity of *E. obliqua* trees to shed branches did not vary between forest types. When a branch is shed, the site of injury is occluded by the tree. However, if heartwood damage exists then the tree is less successful and a hollow develops. The diameter of the branch that is shed influences the probability of hollow formation (Marks *et al.*, 1986; Gibbons, 1999). Dead branch index, as defined in the present study, is not only a function of the abundance of dead branches in the tree, but also of the size of the dead branches. A tree with a greater dead branch index is likely to have a number of large dead branches which, once shed, are likely to result in the successful production of tree hollows. The amount of dead wood in the tree is also likely to be an indicator of the amount of stress experienced by the tree and trees which are more stressed are likely to be less efficient at occluding wounds. Therefore, trees with a greater dead branch index are not just likely to have more hollows, but also deeper and larger hollows. The results of such an effect were seen in the present study. The relative importance of dead branch index in predicting both hollow occurrence and abundance increased with hollow size.

The relationship between hollow occurrence and dead branch index was closely mirrored by that of tree diameter. Tree diameter has been shown to be an important predictor of hollow occurrence in numerous studies (Gibbons *et al.*, 2000b; Lindenmayer *et al.*, 2000; Whitford, 2002). However, tree diameter itself is unlikely to be a primary factor influencing hollow formation (Gibbons and Lindenmayer, 2002). An increase in tree girth is usually related strongly to both the age and senescence of a tree (Chapter 3; Gibbons and Lindenmayer, 2002). These factors are more likely to have a direct influence on hollow formation. Yet despite this lack of direct influence, tree diameter is an extremely useful measure, due both to its frequent use in forest inventory and because it is easily standardised which reduces observer bias. The relationship between tree diameter, age and senescence can vary between sites, which affects the size at which trees begin producing hollows (Gibbons and Lindenmayer, 2002, p 43).

As trees age, the rate of growth generally slows down (Rayner, 1992; Banks, 1993), the tree becomes less able to occlude wounds effectively (Jacobs, 1955), is more vulnerable to decay and is more likely to have been exposed repeatedly to disturbance events that promote the formation of hollows (Gibbons and Lindenmayer, 2002). However, the results of this study indicated that the strength of the relationship between tree age and hollow occurrence decreased with both hollow size and abundance, corresponding with a relative increase in the importance of tree diameter and dead branch index. Furthermore, the relationship between tree age and hollow abundance was not linear, but plateaued. This plateau effect meant that

tree age was an important predictor of hollow presence or absence, but once hollows were known to occur, the size and amount of dead wood in the tree became a more effective predictor of hollow abundance. It therefore seems that *E. obliqua* trees have an age at which they become less successful at occluding wounds. Once this age is reached, the environment in which the tree is growing (e.g. disturbance frequency, wind velocity) is likely to have a greater influence on hollow formation, thereby influencing tree diameter and dead branch index.

The number of stem wounds in a tree can be roughly assessed by visually inspecting the standing tree. Counts of hollows conducted in this manner have been used as estimates of hollow abundance (Lindenmayer *et al.*, 2000; Munks *et al.*, in press). However, not all hollows are easily detected and so some will be overlooked during such surveys (Chapter 4). Therefore these hollow counts should be used as an index rather than an actual count of hollow abundance. The ease with which hollows are detected will vary with the density of the forest and canopy and the size of the tree – factors which are also related to forest type. Yet despite such imperfections in the technique, pre-fall hollow counts appear to be the best method for estimating both hollow presence and abundance. In addition, the more hollows that are observed in a standing tree, the more stress and damage that the tree is likely to have undergone. More stress and damage means that such trees will be more likely to have not just more, but larger hollows.

Recommendations for selecting hollow-bearing trees for retention

In terms of choosing an effective method for identifying hollow-bearing trees for retention in production forests, the simple Classification Tree provides an adequate model. The Classification Tree provided similar predictive ability to the more complex Random Forest and successfully identified the most important variables. Furthermore, the 'survey' models provided equal or superior predictive ability to the 'causal' model. Thus the Classification Tree 'survey' models will provide sufficient predictive ability to be used for identification of habitat trees in production forests. It should be noted, however, that greater accuracy was achieved for predicting hollow presence than absence for total and small hollows, while the reverse was true for large hollows. This greater accuracy is most likely due to the relative prevalence of smaller hollows in the trees examined. The majority of trees examined contained at least one hollow, but relatively few contained a large hollow. The presence of a large hollow was found to be one of the most important predictors of whether a tree would be used by fauna or not (Chapter 6). In the present study it was found that trees that were predicted to have a large hollow when using the survey method were found to not actually have a large hollow 40% of the time.

The inclusion of pre-fall hollow counts was found to produce the models with the greatest accuracy for both the hollow presence and hollow abundance models. Therefore estimates of hollow abundance should be used to identify habitat trees whenever possible. However, forest managers are often looking for a rapid method of assessing trees and surveys of hollows in standing trees are rarely a standard inventory technique. Therefore the utility of the models which include pre-fall hollow counts is minimal for making stand-level assessments or maps of the availability of the hollow resource. Under these circumstances, tree diameter can be a useful substitute without resulting in a severe loss of predictive ability. There was only 1% difference in the misclassification rates for the two models predicting the presence of large hollows (meaning 18.6% of trees expected to have large hollows are likely to be misclassified when using the tree diameter model and 17.5% when using the pre-fall hollow counts model). The improvement was similarly marginal when predicting hollow abundance (Table 5.8). However, the tree diameter model for hollow presence only predicted trees to have large hollows at low altitudes. Given that habitat will be required by fauna at all altitudes, this model is obviously of limited use. If the branch of the Classification Tree involving altitude is disregarded, the misclassification rate of the large hollow model increases to 50%. It is therefore apparent that whenever possible, pre-fall hollow counts and tree diameter should be used concurrently to identify suitable habitat trees, possibly with an additional assessment of the amount of dead wood in the tree.

In some areas of Australia, the prescriptions for habitat tree retention vary between forest types and geographical locations (Wayne *et al.*, 2006) to reflect differences in the age and size at which trees produce hollows suitable for use by fauna. Although a significant difference in hollow abundance was found for trees in the different broad forest types examined, this occurred largely because the trees studied in wet forest were larger and older than their dry forest counterparts. The greater size and age of trees found in wet forest it likely to have occurred as a result of the harvesting or disturbance history in the area. Most dry forest in Tasmania has been repeatedly harvested and burnt since European colonisation. Harvesting in wet forest has historically occurred to a lesser extent and burning is naturally rarer. The results of the current study suggest that the current density of hollows in dry forest areas is less than that of wetter areas due to the younger overall age of the trees that has resulted from repeated harvesting events. This reinforces the need for effective management of animal habitat in production forest areas.

Our results imply that trees of similar ages should be retained in the forest types considered. There was a slight difference in growth rates between the forest types (Chapter 3), which resulted in a very slight difference in the size at which most trees produce hollows. Although this difference was not detected in the predictive models, it is recommended that the size of

trees to be retained in wet forest areas should be slightly larger than that of trees retained in dry forest areas.

Conclusion

The variables found to be most important in this study for predicting hollow presence or abundance in *E. obliqua* trees in dry and wet forest in Tasmania were the number of hollows seen in a tree while it is standing, tree age, diameter and the amount of dead wood the tree contains. Significantly more hollows were found in wet forest than dry, which was largely a result of wetter forest trees being older, larger and with more dead wood. The rate at which trees produced hollows was not found to vary between the forest types examined. The reason why the rate of hollow formation did not differ between the forest types is most likely because it is the shedding of branches which contributed most to hollow formation and the rate of branch shedding appeared to be constant between the forest types examined. However, trees grow slightly more slowly in dry forest than wet. Therefore, the size at which trees produce hollows is slightly greater in wet forest than dry. The reason the predictive models developed did not differentiate between the different forest types is most likely because there was a large degree of overlap in the growth rates between the forest types. This meant that the difference between forest types in the relationship between hollow occurrence and tree diameter was slight enough to not be identified by the models. It is therefore concluded that trees of a similar age should be retained in the different forest types, but that retained trees should be slightly larger in wet forest than dry.

Chapter 6

The use of hollow-bearing trees by vertebrate fauna in wet and dry *Eucalyptus obliqua* forest, Tasmania

Abstract

The loss of hollow-bearing trees in production forest areas can have large impacts on animal populations which rely on them for shelter. This study facilitates the selection of appropriate trees for retention by examining the relationship between site and tree attributes and use of a tree by fauna. The proportion and type of trees that were used by vertebrate fauna in wet and dry *Eucalyptus obliqua* forest in Tasmania, Australia, was examined. Felled trees were searched for hollows and secondary evidence of use by fauna. A combination of Classification Trees, Random Forests and Bayesian logistic regression modelling were used to examine the site and tree attributes most closely related to the use of a tree by fauna. The proportion of trees that were used was at the lower end of the scale found in other areas of Australia, with 28% of hollow-bearing trees examined showing evidence of use. The variables most strongly related to the use of a tree were hollow abundance, tree size and senescence. The likelihood of a hollow being used increased with hollow size, hollow depth in particular. Trees were also more likely to be used in regenerating forest that had some mature trees than any other stand age. Greater use of trees in these stands may be the result of competition for a limited resource (tree hollows) or may indicate a preference by fauna for younger forest provided nesting and roosting sites are available. There was a slight increase in the proportion of trees used with increasing forest wetness, but this most likely represents an increase in the availability of hollows.

The models produced by the current work had low predictive ability. It was therefore recommended that habitat trees be selected according to the presence of large hollows. It is recommended that habitat trees in wet forest areas should be at least 125 cm in diameter and have at least six visible hollows. In dry forest areas a tree should be at least 100 cm in diameter and have at least six visible hollows. Where these prescriptions cannot be met, the next largest tree should be retained.

It was estimated that between 8 and 15 trees per hectare were used by hollow-using fauna. This is significantly more than the rate at which trees are currently being retained within harvesting units in production forests in Tasmania (2 to 3 trees per five hectares).

Introduction

Hollow bearing trees are an important component of a forest ecosystem. They provide nesting and roosting sites for many species of birds, arboreal marsupials, bats, reptiles and invertebrates (Smith *et al.*, 1985; Mackowski, 1987; Webb and Shine, 1997; Gibbons and Lindenmayer, 2002; Bai *et al.*, 2003; Kalcounis-Ruppell *et al.*, 2005). Animals are thought to use hollows because they provide protection from predators, can reduce the energy requirements needed to construct a nest and they have thermoregulatory advantages (Gibbons and Lindenmayer, 2002; Ruczynski and Bogdanowicz, 2005). However, they are a confined space from which it is difficult to escape predators, they can provide an environment conducive to the accumulation of parasites and they can be in short supply (Lindenmayer *et al.*, 1996b; Gibbons and Lindenmayer, 2002).

In some areas, primary excavators, such as woodpeckers, can produce cavities that can also be used by secondary cavity-nesters (Remm *et al.*, 2006). However, woodpeckers are selective in their choice of tree and forest (Jackson and Jackson, 2004) and do not occur in all areas where hollows are required. For instance, there are no primary hollow excavators in Australia and hollows are instead formed by fire, fungi and termites (Mackowski, 1984; Inions *et al.*, 1989; Gibbons and Lindenmayer, 2002). The time required for a hollow to develop is generally much longer than the time between timber harvesting events in production forests (Lindenmayer *et al.*, 1990b; Ball *et al.*, 1999; Gibbons, 1999). To try and maintain a continual supply of hollows for fauna, forest managers require that hollow-bearing trees are retained within a harvesting unit. Current guidelines in Australia for the number and type of hollow-bearing trees to be retained vary both with geographic location and forest type (Wayne *et al.*, 2006). Research in mainland Australian forests on the types of trees that have hollows or are being selected for use by vertebrate fauna has informed the development of these guidelines in many areas. The use of a particular tree by fauna or the density of animals in a particular area has been related to the following site, tree and hollow attributes: understorey, forest structure, topography, altitude, nutrient levels, aspect, slope, disturbance history, distribution of hollows, live or dead tree, tree age, tree size, tree shape, tree species, number of hollows in a tree, decorticating bark index, previous use, hollow dimensions and type of hollow (Braithwaite *et al.*, 1983; Inions *et al.*, 1989; Lindenmayer *et al.*, 1990c; Brigham, 1991; Lindenmayer *et al.*, 1991; Taylor and Haseler, 1993; Kavanagh and Bamkin, 1995; Pausas *et al.*, 1995; Welsh *et al.*, 1998; Gibbons, 1999; Alexander *et al.*, 2002; Wayne, 2005).

In Tasmania, eight bat species, five possums and approximately 29 bird species use hollows to some degree (Chapter 2). This includes several animals listed as threatened in Schedule 3 of the *Tasmanian Threatened Species Protection Act* 1995 (the forty-spotted pardalote, swift

parrot, orange-bellied parrot and masked owl). All of Tasmania's hollow dependent species have priority status under the Tasmanian Regional Forest Agreement (Commonwealth of Australia and State of Tasmania, 1997), meaning that the distribution and abundance of hollows in Tasmanian forests need to be managed carefully. However, in order to ensure management is effective, an understanding of the quantity and attributes of the resource required by fauna is needed. The aims of this paper were firstly to identify the site, tree and hollow attributes that are related to use of a tree or hollow by vertebrate fauna and secondly to determine the proportion of trees that have hollows that have been used by vertebrate fauna and examine any variation between forest types. The results of this work can be used to inform the development of guidelines to assist forest managers in the identification of suitable trees to retain for fauna and the rate at which they should be retained.

Methods

Study sites

Thirty-nine harvesting units in dry or wet *Eucalyptus obliqua* forest (Forest Practices Authority, 2005) were used as study sites (Figure 6.1). The sites were selected by availability in consultation with forest managers from those planned for harvest in State Forest between January 2004 and May 2005. The broad forest categories of 'wet' and 'dry' are based on vegetation mapping produced during the Tasmanian Regional Forest Agreement (Tasmanian Public Land Use Commission, 1996; Harris and Kitchener, 2005). They are two of the most dominant forest types in Tasmania important to the forest industry. What is referred to as 'damp' forest in this paper is a subset of the classification 'dry'. However, as trees in this classification showed characteristics of both wet and dry forest, it was decided to give the subset a separate category. The gradient from dry, through damp to wet forest corresponds to a change in vegetation structure and density, soil nutrient levels, climatic conditions, disturbance history, tree size and age. Dry forest sites were dominated by *E. obliqua* and had an understorey dominated by sedges, heaths or narrow-leaved shrubs. Damp forest sites were dominated by *E. obliqua* and had an understorey dominated by an equal mixture of broad-leaved and narrow-leaved shrubs. Wet forest sites were dominated by, and with a greater than 5% cover, of *E. obliqua* and an understorey containing broad-leaved shrubs, rainforest species, or tall tea-trees.

Figure 6.1 Location of study sites indicating forest type. Squares indicate dry forest, triangles indicate damp forest and circles indicate wet forest.

Data collection

A total of 388 trees was examined, between 6 and 12 per site (196 in dry forest, 76 in damp forest and 116 in wet forest). Information on the characteristics of the sites and trees was collected (see Tables 6.1 and 6.2). These variables were chosen for their anticipated value as predictors of hollow use. The trees examined were felled as part of normal forestry operations. Post-felling, the trees were searched for hollows and hollow characteristics were recorded (Table 6.3). A hollow was defined as a cavity in the tree with a minimum entrance diameter and depth of 2 cm, located at least two metres above the ground. These minimum requirements were selected because an entrance diameter of 2 cm was believed to be the smallest that is generally used by fauna. Birds were observed building nests into shallow scars justifying the use of a depth of 2 cm and 2 m above the ground was considered sufficient to differentiate between arboreal animals and animals located only on the ground. Hollow size classes were based on those used by Gibbons *et al.* (2000b); small: minimum entrance diameter $\geq$ 2 cm, hollow depth $\geq$ 5 cm; medium: entrance $\geq$ 5 cm, depth $\geq$ 10 cm; or large: entrance $\geq$ 10 cm, depth $\geq$ 15 cm. Hollows were searched for evidence of use by fauna and any loose material found in a hollow was collected and returned to the laboratory where it was examined under a dissecting microscope. Evidence suggesting use by fauna included hair, scats, feathers, eggshells, regurgitated pellets, nesting material and obvious rubbing around the entrance of the hollow.

Table 6.1 Site variables recorded.

Variable name	Definition	Model[a]
Forest type	The forest type for the area in which the trees were located was determined using the RFA classification (Forest Practices Authority, 2005) and then reduced to the classes: dry (dominated by *E. obliqua* with understorey dominated by sedges, heaths or narrow-leaved shrubs); damp (dominated by *E. obliqua* with understorey dominated by an equal mixture of broad-leaved and narrow-leaved shrubs); wet (dominated by and with a greater than 5% cover of *E. obliqua*, understorey contains broad-leaved shrubs, rainforest species, or tall tea-trees).	C, S
Basal area	The basal area of each stem with a diameter > 10 cm was measured in a 0.25 ha circular plot located haphazardly within the vicinity of the trees examined. (The actual radius used was dependent on the slope of the site to ensure a horizontal area of 0.25 ha). The results were combined and extrapolated to produce an estimate of the basal area per hectare at each site.	C
Hollow density	The number of hollows observed in standing trees located in the sample area detailed above were tallied and then extrapolated to a per hectare estimate of hollow density.	C
Latitude	A GPS was used to determine the latitude of the sites. This was confirmed using a 1:25,000 a map of the area.	C, S
Longitude	A GPS was used to determine the longitude of the sites. This was confirmed using a 1:25,000 map of the area.	C, S
Altitude	A GPS was used to determine the altitude of the sites. This was confirmed using a 1:25,000 map of the area.	C, S
Aspect	A compass was used to determine the direction directly downslope. The aspects were divided into four categories (N, S, E, W).	C, S
Slope	A clinometer was used to determine the slope of the site in degrees.	C, S
Rock	Parent rock type of the substrate, was obtained from plans developed for the harvesting of the coupe (Forest Practices Plans) and was classified as: granite; dolerite; sedimentary.	C, S
Soil	Soil type as obtained from Forest Practices Plans was classified as: loamy; clayey; sandy.	C, S
Ph	Four soil samples were collected from the top 10 cm. The samples were mixed and the pH was measured using a probe from a 1:5 soil: distilled water solution.	C
Soil conductivity	Soil conductivity was measured as above but using a conductivity meter.	C
Soil nitrogen	Soil was air-dried, ground, sieved and measured for total nitrogen using the Kjeldahl method (Jackson, 1964).	C
Soil phosphorus	As for soil nitrogen, but measured for available phosphorus by acid-fluoride extraction (Jackson, 1964).	C
Topography	The average topography of the site was categorized as: ridge; upper slope; mid-slope; lower slope or gully.	C, S
Canopy cover	The percent cover of the canopy was estimated using a modified Braun–Blanquet index (Mueller-Dombois and Ellenberg, 1974).	C, S

[a] C indicates the variable was used in the 'causal' model and S indicates use in the 'survey' model.

Table 6.1 continued

Variable name	Definition	Model[a]
Understorey cover	The percent cover of the understorey was estimated using a modified Braun–Blanquet index (Mueller-Dombois and Ellenberg, 1974).	C, S
Stand age	The age of the stand in which the trees were found was categorised according to the information from the photographic interpretation (PI) of the vegetation age structure within each plot (obtained from Forestry Tasmania's concise-PI type maps: Stone, 1998) into: (M) Mature; (MR) Mature with regrowth; (RM) Regrowth with mature; (R) Regrowth.	C, S
Annual mean temperature	Annual mean temperature values obtained from ESOCLIM (McMahon *et al.*, 1995) using data on the latitude, longitude and altitude of the site.	C, S
Rain	Annual mean precipitation values obtained from ESOCLIM using data on the latitude, longitude and altitude of the site.	C, S
Radiation	Annual mean radiation values obtained from ESOCLIM using data on the latitude, longitude and altitude of the site.	C, S
Road	The kilometers of road within a 100 m, 300 m and 1000 m radius of the site.	C, S
Reserve	The percentage of land that was found in reserves within a 300 m and 1000 m radius of the site.	C, S
State Forest	The percentage of land that was State Forest within a 300 m and 1000 m radius of the site.	C, S
Private	The percentage of land that was privately owned within a 1000 m radius of the site.	C, S

[a] C indicates the variable was used in the 'causal' model and S indicates use in the 'survey' model.

Table 6.2 Tree variables recorded.

Variable name	Definition	Model[a]
Species	Tree species was determined.	C, S
Dbh	Tree diameter at breast height over bark (1.3 m) measured using a diameter tape (cm).	C, S
Age	Tree age (years) was determined by ring counting when a wood sample was available (growth rates were extrapolated to estimate the age of the missing section if it occurred). For trees with no wood sample a growth model was used. Details are provided in Chapter 3.	C
Shape	A classification of the shape of a tree, based on definitions by Smith and Lindenmayer (1988): (3) Tree with full round crown; (4) Mature tree with major branching; (5) Mature tree with dead branches; (6) Mature tree with dead top but big crown; (7) Mature tree with dead top and small crown; (8) Burnt tree with only epicormic growth.	C, S
Dominance	The relative dominance of each tree was categorized as: (1) dominant; (2) co-dominant; (3) sub-dominant or suppressed.	C, S
Burn damage	The burn damage of the tree was assessed as: (1) no evidence of fire; (2) bark damage; (3) wood exposed and damaged by fire; (4) severe fire damage where the base of the tree forms a bridge.	C, S
Dead branch index	The number of dead branches in each of the following size categories was estimated: (S) Small (5-15 cm); (M) Medium (15-40 cm); (L) Large (> 40 cm). An estimate of dead branch index was obtained by the following formula: DBI = S + 5M + 10L.	C, S
Bark width	The maximum width of the bark (cm) was measured post-felling to accuracy 0.5 cm using a measuring tape.	C, S
Tree height	Tree height was determined using a measuring tape and a clinometer. Measurements were taken about 20 m from the base of the tree. Trees around 30 m in height were most accurately measured in this way (as determined by comparing with felled trees), with taller trees being overestimated and shorter trees being underestimated.	C, S
Crown volume	The volume of the crown (m^3) as estimated using the diameter of the crown north-south and east-west, and the estimated depth of the crown.	C, S
Leaf nitrogen	Five young, mature leaves were collected from each tree in the post-fall survey. They were dried in an oven, then ground, sieved and measured for total nitrogen using the Kjeldahl method (Jackson, 1964).	C
Leaf phosphorus	As for leaf nitrogen, but measured for total phosphorus by mixed-acid digestion and calometric analysis (Allen, 1974).	C
Canopy access	Canopy access is a measure of the quantity of vegetation in close proximity to the tree foliage. One point is scored for each 3 m vertical section of a hollow-bearing tree where surrounding vegetation was < 10 cm from the tree, on either side of the tree. A cumulative score is calculated.	C, S
Trunk access	Trunk access was calculated as above but on the trunk.	C, S
Pre-fall count	The number of hollows observed during the ground-based (pre-fall) survey.	S
Hollow count	The number of hollows (total, small, medium and large) found during the post-fall survey and then corrected for the amount of error (see Chapter 4).	C

[a] C indicates the variable was used in the 'causal' model and S indicates use in the 'survey' model.

Table 6.3 Hollow variables recorded.

Variable name	Definition
Branch order	The order of the branch where 1 is the trunk, 2 is a first order branch, 3 is the second order branch and 4 is third and greater order branches.
Min entrance size	The minimum entrance diameter (cm) of the hollow.
Hollow depth	The depth of the hollow (cm).
Hollow form	A classification of hollow type: (1) hollow goes horizontally in; (2) hollow angles down into the branch; (3) hollow extends both above and below the hollow entrance; (4) crescent shaped hollow in the collar of wood at the base of a branch; (5) very open hollow showing almost completely inside.
Hollow shape	The shape of the inside of the hollow: even; variable; conical; enlarges.
Entrance shape	A classification of the entrance shape as: round; oval; irregular.
Hollow height	The height of the hollow above the ground as measured post-felling (m).
Branch aspect	The aspect in which the hollow was facing on the branch (°): (1) nnw to ne; (2) ene to se; (3) sse to sw; (4) wsw to nw; (5) up; (6) down.
Tree aspect	The side of the tree on which the branch containing the hollow was located: (1) nnw to ne; (2) ene to se; (3) sse to sw; (4) wsw to nw; (5) up.
Hollow live or dead	Whether the wood at the entrance of the hollow is live, dead or a combination.
Internal texture	A classification of the internal surface texture of the hollow: (1) smooth; (2) intermediate; (3) jagged.
External texture	A classification of the surface texture of the hollow entrance: (1) smooth; (2) intermediate; (3) jagged.
Branch diameter	The diameter of the branch on which the hollow was found (cm).
Hollow type	Classified as: (1) hole; or (2) scar.

Using hairs to identify species is a useful technique, but can vary in the level of accuracy achieved. In a study done by Lobert *et al.* (2001), only 54-60% of species were identified correctly and without confusion with other species. For the species examined which are also found in Tasmania, the ringtail and brushtail possums were suggested to have questionable reliability for hair identification, the sugar glider was listed as both reliable and unreliable (presumably due to the differences between hair identifiers) and the eastern pygmy possum was listed as unreliable. The accuracy of the technique is believed to increase with the competence and experience of the person using it. In the present study, all hairs collected were sent to an experienced person (B. Triggs) for identification to species level where possible. However, because the errors associated with the technique can be substantial when identifying hair samples to species, the indirect evidence collected was largely used to only confirm tree use or non-use.

The method of felling trees to determine use by fauna has some limitations (Chapter 4). When trees are felled they smash to varying degrees and it is likely that the hollows are the areas of weakened wood that are more susceptible to smashing. No method for correcting for this error was available. However, even when a hollow was smashed, the position of the hollow could often be identified by a piece of hollow edge or collar. Consequently, the loose material could still be searched for faunal evidence. In addition, while a large number of hollows were not found in the post-fall survey in larger trees (Chapter 4), there were still a lot of hollows found in these trees that could be examined. This means that although it is harder to say definitively that a tree is not being used by fauna, due to hollows not being found, it can be said when a tree is being used. Trees were considered to be 'used' if even one hair or feather was found. It is likely that some of these hollows were not actually used for nesting.

Data analysis

Modelling tree use

Three modelling techniques were used to explore the relationship between site and tree variables and tree use by fauna. Classification Trees were initially selected for several reasons. The primary advantage of this method is that the output is easily interpreted and can readily be translated into management guidelines. The second major advantage of this method is the lack of assumptions that are required and the robustness of the model for dealing with missing data (De'Ath and Fabricius, 2000; Hastie *et al.*, 2003). Classification Trees were constructed using the 'mvpart' package (Therneau *et al.*, 2006) using the statistical program R (R Development Core Team, 2006) and the trees were cross-validated by site number as was done in Chapter 5. However, the recommended method for identifying the optimal Classification Tree size (De'Ath and Fabricius, 2000) did not work for the Classification Trees examined in the present study. For the current data sets, the null model (no splits) was identified as the best model, with a slight increase or decrease in relative error occurring for the next smallest Classification Tree. However, the smallest Classification Tree was within one standard error of the null model and this is believed to be sufficient to justify examination of the models.

Classification Trees do not always have strong predictive ability and Random Forests are recommended to be used in conjunction (Prasad *et al.*, 2006) using the 'randomForest' library in R (Liaw and Wiener, 2002). While Random Forest models are not intuitive, the output does provide a useful indication of the amount of predictive ability that was lost by using a simple Classification Tree and the relative importance of the input variables by means of the Gini index. The Gini index is a measure of the total decrease in node impurity resulting from the splitting of the variable, averaged over all trees. Node impurity refers to

the proportions of responses in each category of the dependent variable that occur at each node or 'branch' (De'Ath and Fabricius, 2000). Random Forests do not possess the same ability as the Classification Trees for dealing with missing data. When data were missing for a particular tree, that tree was excluded from the Random Forest analysis. An assessment of basal area and hollow density (Table 6.1) was not done for nine of the sites considered. Consequently, the Random Forest was initially done using all input variables. The Gini values were examined and if either of these variables contributed little to the model, they were removed and the analysis was redone with the larger data set. One thousand trees were grown for each Random Forest.

Due to uncertainty in the results of the Classification Tree model, the analysis was repeated using a regression style analysis. Bayesian logistic regression models with uninformative priors were done in WinBUGS 1.4 (Spiegelhalter *et al.*, 2003). Selection of the most important variables relating to use or non-use was done using a forward stepwise approach, investigating single and two-way interactions. All continuous predictor variables were standardised to reduce autocorrelation between successive samples (i.e. the mean was subtracted from the data, which was then divided by the standard deviation). The initial 1,000 samples were discarded as a 'burn in'. The following 10,000 samples were used to calculate the Deviance Information Criterion (DIC) which was used to select the best model. DIC values can be interpreted in a similar way to AIC values (Burnham and Anderson, 2002), where a difference of less than two indicates a lack of difference in the models and a difference of three or more indicates that the model with the smaller DIC value is superior (McCarthy, 2007). To determine the degree to which variation occurred between sites, that could not be attributed to the site attributes measured, 'site number' was added as a random factor. For the final model (excluding site number), 100,000 samples were used to calculate the mean, standard deviation, 2.5^{th} and 97.5^{th} percentile of the coefficients. The percentiles represent a 95% Bayesian credible interval. To evaluate the fit of the model, all data used to produce the model were used to predict whether a tree was used by fauna or not. The number of informative predictions was examined, where a prediction was considered to be informative if the 95% credible interval did not span from zero to one. In addition, the fit was assessed by the following formula, which estimates the proportion of deviance explained by the fitted model:

$$GOF = (Dhat_1 - Dhat_0) / Dhat_0 \times 100$$

GOF = Goodness of fit measure (percentage of variance in the data explained by the variables)

$Dhat_1$ is the Dhat value produced by WinBUGS for the fitted model. (Dhat is a point estimate of the model deviance: McCarthy, 2007).

$Dhat_0$ is the Dhat value where all parameters are removed except an uninformative constant

In order to compare the predictive ability of the three modelling techniques, rates of misclassification for each were determined. For the regression model, a predicted value of more than 0.5 was taken to indicate that a tree was likely to be used by fauna and smaller values indicated it was unlikely to be used.

Two sets of models were developed using each of the three methods; Classification Trees, Random Forests and Bayesian regression modelling. The first included all variables that were thought to potentially contribute to the use of a tree or hollow in an ecologically meaningful way (Table 6.1 and 6.2). This set of models will be referred to as the 'causal' models. The second included only those variables that are easily measured in the field and are therefore more practical indicators of the potential use of a tree by fauna. These models will be referred to as the 'survey' models. Two alternative 'survey' Classification Trees were produced as they explained similar amounts of variability in the data (see results).

Rates of tree use and forest type

Examining the rate at which trees in a stand are used by fauna is useful for assessing the adequacy of tree retention prescriptions in production forest areas. In order to do this, the diameter and abundance of hollows was assessed for all trees greater than 10 cm in diameter in a 0.25 ha plot at 33 sites (13 dry forest sites, 9 damp forest sites and 11 wet forest sites). The plot size used was selected due largely to time constraints. A pilot study was done (unpublished data) to examine the accuracy of this plot size for measuring the stand structure of the forest. It was estimated that when using a 0.25 ha area, 60% of sites sampled would give an estimate of tree density within 20% of the actual value for trees of high densities (> 100 trees/ha). For trees of low density (10-20 trees/ha), fewer than 50% of sites would give an estimate of tree density within 30% of the actual value. Therefore the small plot size used may cause substantial error in establishing the density of some tree sizes, particularly the larger trees.

All but four of these sites were used in the previous analyses. The results of the surveys were multiplied by four to produce per hectare estimates of tree and hollow density. The first step for establishing the difference in rates of hollow use between forest types was to establish if there was a difference in hollow occurrence between the forest types. ANOVA was used to assess if a difference occurred between forest types in the density of hollows or hollow-bearing canopy trees when considering all hollows or holes only (i.e. not fire scars).

In order to assess the rate at which hollows were used in the different forest types, the rates of hollow use for the different forest types in relation to tree diameter and tree age was

examined graphically for the 388 trees used in the modelling process. Three different techniques were then used to make estimates about the number of trees likely to be used by fauna in a one hectare area of *E. obliqua* forest. Firstly, the total proportion of trees used in a particular forest type was multiplied by the number of trees at least 50 cm in diameter that were estimated to occur in a one hectare area. These estimates of tree density were obtained from the 0.25 ha surveys mentioned above. Secondly, the two 'survey' Classification Tree models produced in the results section of this paper were used to predict whether a particular tree would be used by fauna. The misclassification rates of the model were then used to correct the number of trees predicted to be used. This technique is best expressed by an example. Take a hypothetical situation where ten trees in a one hectare area were predicted by the model to be used by fauna and there were a further 25 trees that were at least 50 cm in diameter but were predicted to not be used by fauna. In the model it was found that 65.6% of the trees predicted to be used by fauna were correctly classified. Of the trees predicted to not be used by fauna, 18.8% were misclassified. The number of trees at the site being used by fauna is estimated as follows:

$$\text{Number of trees used} = (0.656 \times 10) + (0.188 \times 25)$$

The results of this process were summarized (mean and standard deviation) for all sites within a particular forest type.

Hollow use

For some of the hollows examined it was not possible to collect measurements for all the hollow attributes indicated (Table 6.3) due to varying degrees of smashing. Consequently, the sample sizes differ for some analyses and it is indicated in the text where this occurs. Attempts at modelling the variables relating to the use of a particular hollow were done using all independent variables in Table 6.1, 6.2 and 6.3 (with the exception of pre-fall hollow counts). However, problems were encountered with production of the Classification Trees; the Tree produced was too large to be considered realistic, the relative error was much greater than that of the null model and the Tree did not appear to be stable. Problems also arose when producing a Bayesian logistic mixed-effect regression model; the data was highly autocorrelated, despite standardisation of all variables. Although both models indicated that hollow depth was of primary importance, the models were considered too poor to be presented. As a model predicting the use of a particular hollow was not required, an exploration of the relative importance of the independent variables (as done by the Random Forest) was considered sufficient. A summary of the nesting hollows found for each of the species identified is also provided.

Results

Three hundred and nineteen of the 388 trees examined had a hollow (249 had a small hollow, 227 had a medium hollow and 93 had a large hollow). Eighty-eight (27.6%) of the hollow-bearing trees showed evidence of use by fauna, as did 74 (29.7%) of the trees with small hollows, 78 (34.4%) of the trees with medium hollows and 49 (52.7%) of trees with large hollows. There was little variation between forest types in the proportion of trees that showed evidence of use by fauna (21.2%, 23.4% and 24.8% of trees in dry, damp and wet forest respectively). Of the trees that contained at least one large hollow, 23 of the 45 trees (51.1%) were used in dry forest, six of the ten (60%) were used in damp forest and 20 of the 38 (52.6%) were used in wet forest.

Tree use 'causal' models

The best 'causal' Classification Tree predicting use of a tree by fauna included the number of large hollows in the tree and stand age (Figure 6.2). However, the model had only slightly better predictive ability than the null model (misclassification of null = 0.227, model = 0.180). The model indicated that trees with no large hollows were unlikely to be used. Those with more than five large hollows were very likely to be used. For those with between one and five large hollows, the trees were more likely to be used when found in regrowth forest with a mature component.

The best 'causal' Bayesian model (Table 6.4) showed a similar pattern to that of the 'causal' Classification Tree model. The likelihood of a tree being used by fauna increased with the number of hollows occurring in the tree. For stand age, the coefficients for all categories except 'regrowth with mature' included zero in their 95% credible intervals. This supports the result of the Classification Tree that trees are more likely to be used in this type of forest stand. The main difference between the Bayesian and Classification Tree model was the inclusion of the square root of dead branch index as an additional predictive variable. This relationship indicates that the more dead branches there are in the tree, the more likely it is that the tree will be used by fauna. The random factor site did not improve the fit of the model as measured by the DIC value. The GOF value when site was removed was 0.192 indicating the model explained under a fifth of the variability in the data. All of the predictions were uninformative (had a 95% credible interval that spanned from zero to one).

The results of the Random Forest confirm the result that hollow abundance was the variable most strongly related to tree use, irrespective of hollow size (Table 6.5). However, the results also indicated that the likelihood a tree will be used by animals is related to the size, age and senescence of the tree. The only indication of this given in the previous two models was the

inclusion of dead branch index in the Bayesian model. It is possible that other variables such as tree diameter did not occur in the previous models because they were strongly related to hollow abundance (Chapter 5). The variable conspicuous by its absence in the Random Forest model was stand age. The misclassification rates were similar for all three models, being very low for predicting non-use but having high levels of error for predicting use of a tree by fauna (Table 6.6).

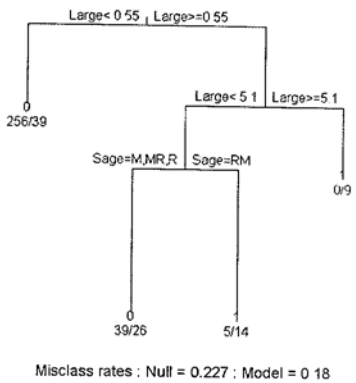


Figure 6.2 Results of the Classification Tree 'causal' model for whether a tree showed evidence of being used by fauna or not. 'Large' indicates the number of large hollows in the tree and 'Sage' indicates the age of the stand. The vertical length of the branch represents the relative strength of that split. The zero or one at the end of each branch represents whether that leaf predominantly represents tree non-use or use respectively. The numbers below that represent the number of samples in the current data set that were found at that leaf, where the number on the left side of the slash is the number of samples in which trees showed no signs of use by fauna and the number on the right of the slash is the number of samples in which the tree did show signs or use. For the Misclass rates, the Null rate is the error when the rule 'go with the majority' is used. The Model rate is the misclassification rate found from using the stated model.

Table 6.4 Coefficients (mean, standard deviation, 2.5th and 97 5th percentile) of the explanatory variables included in the best 'causal' Bayesian logistic regression model for whether a tree was used by fauna or not. [a]

Variable	mean	sd	2.5%	97.5%	DIC [b]
Intercept	-1.456	0.222	-1.902	-1.037	415.0
Hollow abundance	0.059	0.015	0.031	0.089	358.3
Sqrt (Dead branch index)	0.200	0.059	0.085	0.316	351.4
Stand age: mature	0				344.9
Stand age: mature with regrowth	-0.697	0.407	-1.515	-0.081	
Stand age: regrowth with mature	0.628	0.333	0.024	1.284	
Stand age: regrowth	-0.883	0.588	-2.094	0.2131	
Site					346.0

[a] The values presented are for the model excluding the random factor 'site'.

[b] The DIC values are the progressive values attained with the addition of each new variable.

Table 6.5 The ten 'causal' variables most strongly related to tree use as indicated by the Random Forest when 1000 trees were grown.[a]

Variable	Gini [b]
No. hollows	10.3
No. large hollows	9.5
No. medium hollows	8.3
No. small hollows	7.8
Dead branch index	6.9
Tree diameter	6.8
Tree age	5.8
Crown volume	4.8
Tree height	4.8
Tree shape	3.7

[a] Variables removed from the data set for this analysis were: basal area and hollow density.

[b] The values presented are the total decrease in the Gini index (node impurity).

Table 6.6 Comparing the misclassification rates for the Classification Tree, the Bayesian regression analysis and the Random Forest when using the 'causal' data set. [a]

			Predicted result		Classification error
			Non-use	Use	
Classification Tree		Non-use	295	5	0.017
		Use	65	23	0.739
Bayesian Regression	True	Non-use	286	14	0.047
	result	Use	64	24	0.727
Random Forest		Non-use	231	22	0.087
		Use	53	23	0.697

[a] The sample size is slightly smaller for the Random Forest because the model included some variables with missing data and so these data points needed to be omitted.

Tree use 'survey' models

There were two competing Classification Trees for the 'survey' data set (Figure 6.3). The first included the number of hollows seen in the pre-fall survey and stand age. Reminiscent of the 'causal' model, it indicated that trees were likely to be used if more than two hollows were observed and the tree was located in regenerating forest with a mature component. The second included tree diameter, crown volume and site altitude. Trees smaller than 115 cm in diameter were unlikely to be used by fauna. Larger diameter trees were likely to be used if they had a large crown. Trees large in diameter but with a small crown were likely to be used if they were found at high altitude. It should be noted that tree diameter could also be replaced by dead branch index and provide a roughly equivalent model. Tree diameter was used because it is more practical in the field. While the overall error for the second model was smaller than that of the first, the misclassification of trees being used by fauna was greater (Table 6.9).

The best Bayesian 'survey' model was quite different to the Classification Tree 'survey' model. It included an interaction between tree shape and the number of hollows seen in the pre-fall survey, the square root of dead branch index and stand age (Table 6.7). However, tree shape 3 (Table 6.7) was the only shape where zero was not included in the 95% credible interval. This indicates that for trees of this shape, the likelihood of being used by fauna increased with the number of hollows seen in the pre-fall survey. The effect was not as strong, or did not exist for other tree shapes. As for the 'causal' model, there was a positive effect of dead branch index and trees found in regrowth with mature forest were more likely to be used by fauna. However, there was also an additional but negative effect of being in mature forest with some regrowth trees. The GOF for this model was 0.22. Eight point six per cent of predictions were informative (the 95% credible interval did not span from zero to one). Of these, 93.9% were for trees that were not used by fauna, indicating a greater ability to predict tree non-use than use.

The results of the Random Forest 'survey' model were similar to that of the Random Forest 'causal' model, relating tree use to tree size and senescence (Table 6.8). The number of hollows seen in the standing tree is an imperfect indication of hollow abundance (Chapter 4) but was identified as a useful variable for predicting tree use. Again, stand age was not identified as being an important variable and nor was site altitude. It should be noted however that despite the labels for the stand age categories, regrowth forest had more large and old trees than regrowth with mature forest (Figure 6.4). This has occurred because the distribution of regrowth and older trees in a stand can be quite patchy and the area examined in the four harvesting units classified as being regrowth generally contained some older trees. In comparing the three models, the Bayesian regression analysis appears to have lower

misclassification rates than either the Classification Tree or the Random Forest (Table 6.9). The misclassification rates for the latter two models were greater than for the equivalent 'causal' models.

a)

b)

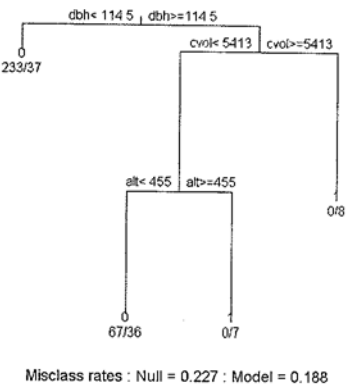


Figure 6.3 Results of the Classification Tree 'survey' model for whether a tree showed signs of being used by fauna or not when (a) including pre-fall counts and (b) excluding pre-fall counts. 'Hollowpre' indicates the number of hollows seen in the pre-fall survey. 'Sage' indicates the age of the stand (M = mature, MR = mature with regrowth, RM = regrowth with mature and R = regrowth). 'Dbh' indicates the tree diameter. 'Cvol' indicates the volume of the crown. 'Alt' indicates the altitude of the site. The vertical length of the branch represents the relative strength of that split. The zero or one at the end of each branch represents whether that leaf predominantly represents tree non-use or use respectively. The numbers below that represent the number of samples in the current data set that were found at that leaf, where the number on the left side of the slash is the number of samples in which trees showed no signs of use by fauna and the number on the right of the slash is the number of samples in which the tree did show signs or use. For the Misclass rates, the Null rate is the error when the rule 'go with the majority' is used. The Model rate is the misclassification rate found from using the stated model.

Table 6.7 Coefficients (mean, standard deviation, 2.5th and 97 5th percentile) of the explanatory variables included in the best 'survey' Bayesian logistic regression model for whether a tree was used by fauna or not. [a]

Variable	mean	sd	2.5%	97.5%	DIC [b]
Intercept	-1.066	0.225	-1.519	-0.634	408.0
Pre-fall count					378.9
Shape 3: Pre-fall count	0.507	0.110	0.304	0.734	367.3
Shape 4: Pre-fall count	0.114	0.085	-0.047	0.286	
Shape 5: Pre-fall count	0.089	0.133	-0.168	0.358	
Shape 6: Pre-fall count	0.031	0.049	-0.066	0.128	
Shape 7: Pre-fall count	-0.008	0.080	-0.172	0.145	
Sqrt (Dead branch index)	0.208	0.062	0.088	0.330	353.4
Stand age: mature forest	0				339.5
Stand age: mature with regrowth	-1.114	0.455	-2.038	-0.253	
Stand age: regrowth with mature	0.871	0.346	0.199	1.558	
Stand age: regrowth	-0.570	0.567	-1.732	0.496	
Site					339.4

[a] The values are for the model excluding the random factor 'site'. n = 383 because trees of shape 8 were removed due to such low numbers.

[b] The DIC values are the progressive values attained with the addition of each new variable.

Table 6.8 The ten 'survey' variables most strongly related to tree use as indicated by the Random Forest when 1000 trees were grown.

Variable	Gini [a]
Pre-fall count	12.0
Tree diameter	11.8
Dead branch index	11.8
Crown volume	8.5
Tree height	7.6
Tree shape	5.6
Bark width	4.4
Burn damage	3.8
Canopy access	3.8
Trunk access	2.5

[a] The values presented are the total decrease in the Gini index (node impurity).

Table 6.9 Comparing the misclassification rates for the Classification Tree, the Bayesian regression analysis and the Random Forest when using the 'survey' data set. [a]

			Predicted result		Classification error
			Non-use	Use	
Classification Tree a	True result	Non-use	289	11	0.037
		Use	67	21	0.761
Classification Tree b		Non-use	300	0	< 0.001
		Use	73	15	0.830
Bayesian Regression		Non-use	277	20	0.067
		Use	56	30	0.651
Random Forest		Non-use	231	22	0.087
		Use	63	13	0.829

[a] The sample size is slightly smaller for the Bayesian regression because trees of shape 8 were removed for the Random Forest because the model included some variables with missing data and so these data points were omitted.

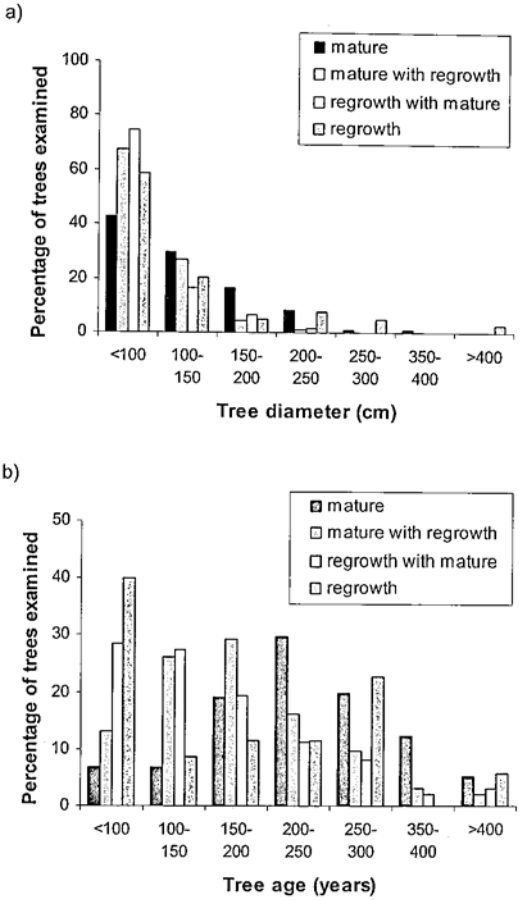


Figure 6.4 The percentage of trees examined found in each (a) tree diameter and (b) tree age category in relation to stand age.

Rates of tree use

Between 12 and 64 (mean 40.1 ± 15.7 s.d.) hollow-bearing trees per hectare were estimated to occur in the 33 sites examined. There was no significant difference between the different forest types in the density of hollow-bearing canopy trees either when considering all hollows ($F = 0.766$, $p = 0.474$, $df = 2$) or holes only ($F = 0.444$, $p = 0.646$, $df = 2$, Figure 6.5a). However, there was a trend for hollow density to increase with forest wetness, significant at the 0.06 level ($F = 3.3$, $p = 0.053$, $df = 2$, Figure 6.5b). This indicates that the abundance of hollows found on a particular tree increased with forest wetness.

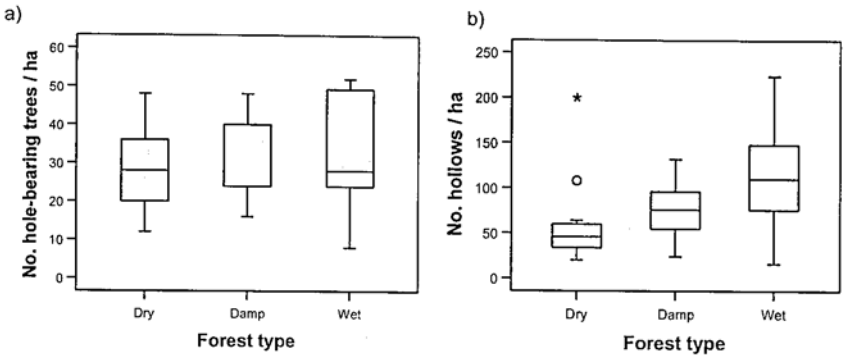
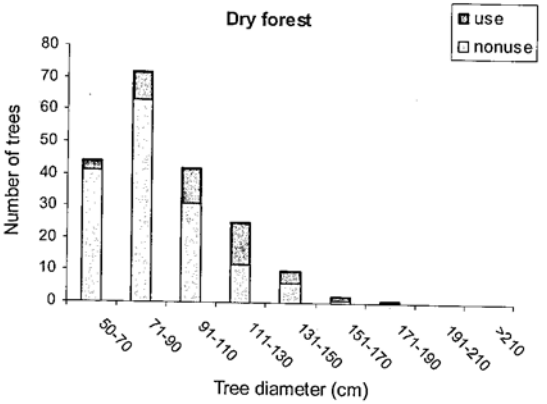


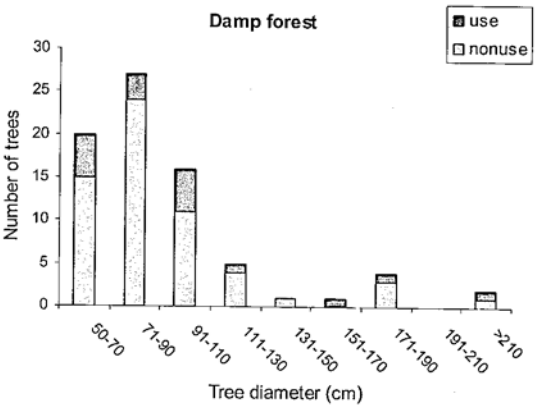
Figure 6.5 The relationship between forest type and (a) the number of canopy trees per hectare that contained a hole (scars excluded) and (b) the number of hollows per hectare. Observations of hollows were done in a 0.25 ha plot and then extrapolated to one hectare. The box length is the interquartile range. The circles are outliers with values between 1.5 and 3 box lengths from the upper or lower edge of the box. The stars are outliers with values more than 3 box lengths from the upper or lower edge of the box. n = 12 for dry forest, 7 for damp forest and 10 for wet forest.

For the trees examined in this study, it was found that some small trees showed evidence of use by fauna in both dry and damp forest (Figure 6.6a and b). In comparison, no trees less than 110 cm in diameter were found to be used by fauna in wet forest and the likelihood of use increased with diameter (Figure 6.6c). In contrast, the three forest types showed a similar trend towards an increase in the proportion of trees being used with tree age (Figure 6.7), although the abundance of the older trees increased with forest wetness.

a)



b)



c)

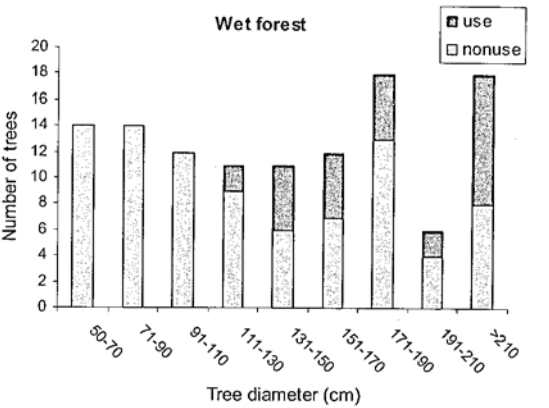
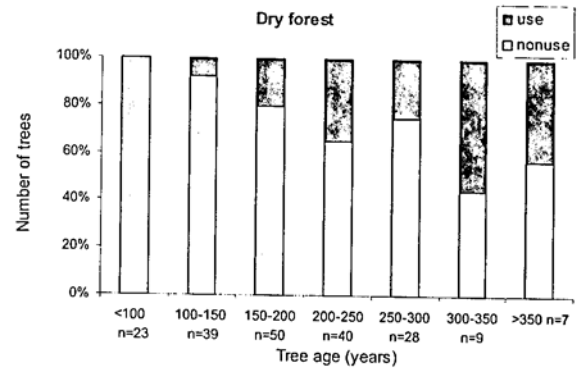
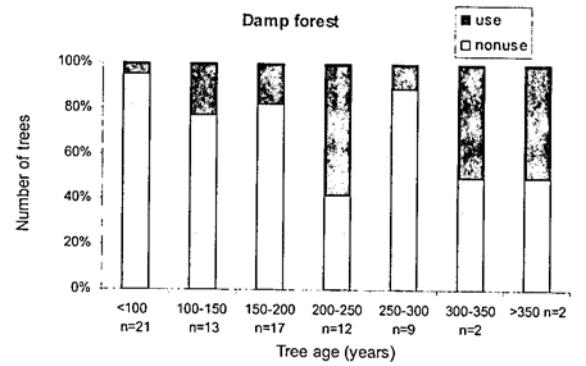


Figure 6.6 The relationship between tree diameter and tree use for (a) dry forest, (b) damp forest and (c) wet forest.

a)



b)



c)

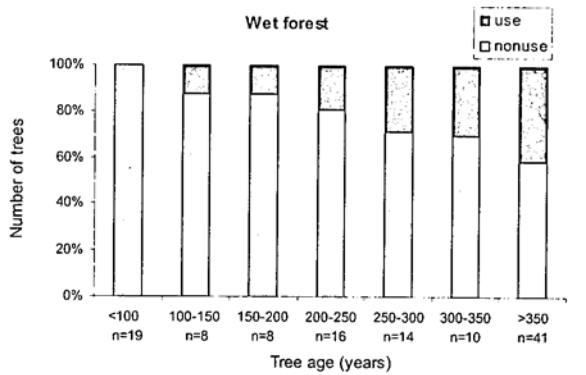


Figure 6.7 The relationship between tree age and tree use for (a) dry forest, (b) damp forest and (c) wet forest.

Estimates of tree use ranged between 8 and 12 trees per hectare in dry forest, 9 and 13 in damp forest and 10 and 15 in wet forest (Table 6.10). All three methods predicted a slight increase in the rate at which trees are used with increasing forest wetness. This corresponds with the increase in hollow availability with forest wetness (Figure 6.5). The estimates made using the predictive Classification Tree models produced in the previous section of this chapter were consistently lower than the estimates made using the total proportion of trees used in that forest type (Table 6.10). Since this latter method was based on the proportion of trees examined that were actually found to be used by fauna, it is expected that these figures lie closer to the true value.

Table 6.10 Estimates of the number of trees that are likely to be used by vertebrate fauna in a one hectare area according to the different forest types (mean and s.d.).

Technique used	Forest type		
	Dry	Damp	Wet
Extrapolation for all trees ≥ 50 cm in diameter [a]	11.7 ± 3.0	12.7 ± 4.0	14.8 ± 7.0
Use model from Figure 6.3a [b]	8.3 ± 2.9	9.5 ± 3.3	10.4 ± 6.1
Modification of use model from Figure 6.3b [c]	8.4 ± 2.6	9.3 ± 2.9	12.8 ± 6.1

[a] This technique multiplied the number of trees per hectare that were at least 50 cm in diameter by the rates of hollow use found in the respective forest types (0.212 for dry forest, 0.234 for damp forest and 0.248 for wet forest).

[b] This technique used the model from Figure 6.3a, and multiplied the misclassification rates by the number of the trees per hectare that met the requirements of each node of the model.

[c] This technique used a modification of the model from Figure 6.3b. No data on crown volume was collected in the plots examined and therefore the misclassification rates for trees at least 115 cm in diameter and those smaller were used to estimate the proportion of trees in a hectare used by fauna. Trees were classified as having a high likelihood of being used by fauna (0.43) if they were at least 115 cm in diameter and a low likelihood (0.14) if they were smaller.

Hollow use

One hundred and twenty-seven (5.4%) of the 2,357 hollows examined showed signs of use by fauna. Twenty-three of the 756 small hollows examined (3.0%) were used, 44 of the 653 medium hollows (6.7%) and 38 of the 156 large hollows (24.4%). Seven hundred and ninety-two hollows were not assigned a size due to varying degrees of smashing. Of these, 22 (2.8%) showed evidence of use by fauna. Ninety-eight (7.1%) of the 1,381 hollows found to be at least 2 cm in entrance diameter and 5 cm deep showed evidence of use (7.7% if no scars). Only 1.3% of hollows (22 out of 1661) less than 15 cm deep showed evidence of use by fauna in comparison to 14.9% (81 of 545) of deeper hollows. Of the 99 hollows which showed evidence of use and hollow entrance width and depth were measured (the rest were not due to varying degrees of smashing), the hollows were on average 5.3 (± 9.7 s.d.) times as deep as they were wide (with a median value of 2.5)

For the Random Forest model, a number of variables were removed from this analysis (basal area, hollow density, bark width, canopy access, trunk access and soil conductivity). This

was done because unlike Classification Trees, Random Forests have no easy way to deal with missing data. Consequently variables with a large amount of missing data that were shown to not contribute much to the predictive ability of the Random Forest were removed to allow a greater sample size. The Random Forest indicated that hollow depth and hollow entrance diameter were the variables most strongly related to hollow use (Table 6.11). Other factors of lesser importance were the diameter of the branch in which the hollow was found and the height of the hollow above the ground. The likelihood of a hollow being used increased with hollow depth, entrance width, branch diameter and decreased with the height of the hollow above the ground (Figure 6.8). The relationships were generally consistent except for branch diameter, where an increase was evident until a diameter of at least 40 cm was reached when it plateaued. The apparent drop in percentage of hollows being used at a branch diameter between 90 and 100 cm is believed to be an artefact of the data and not a real effect. If one more hollow in this size range had been observed to be used, it would have brought the rate of use up to 4.3% which is equivalent to the other branch sizes. The Random Forest model had very high error rates for predicting use of a hollow by fauna (Table 6.12).

Table 6.11 The ten variables most strongly related to hollow use as indicated by the Random Forest when 1000 trees were grown and the following variables were removed: basal area, hollow density, bark width, canopy access, trunk access and soil conductivity.

Variable	Gini [a]
Hollow depth	18.35
Min entrance	13.99
Branch diameter	9.53
Hollow height	7.83
Hollow type	6.13
Trunk aspect	5.54
Branch aspect	5.15
Tree diameter	4.68
Dead branch index	4.40
Tree age	4.20

[a] The values presented are the total decrease in the Gini index (node impurity).

Table 6.12 The misclassification rates for the Random Forest. [a]

		Predicted result		Classification error
		Non-use	Use	
True	Non-use	1653	4	0.002
result	Use	75	11	0.872

[a] Classification Trees can use data even if it has some missing values using the function 'rpart', but Random Forests cannot and so such data was omitted. To improve the sample size, a number of variables were removed: basal area, hollow density, conductivity, hollow shape, hollow type, entrance shape. Their removal had negligible impact on the error of the Random Forest model.

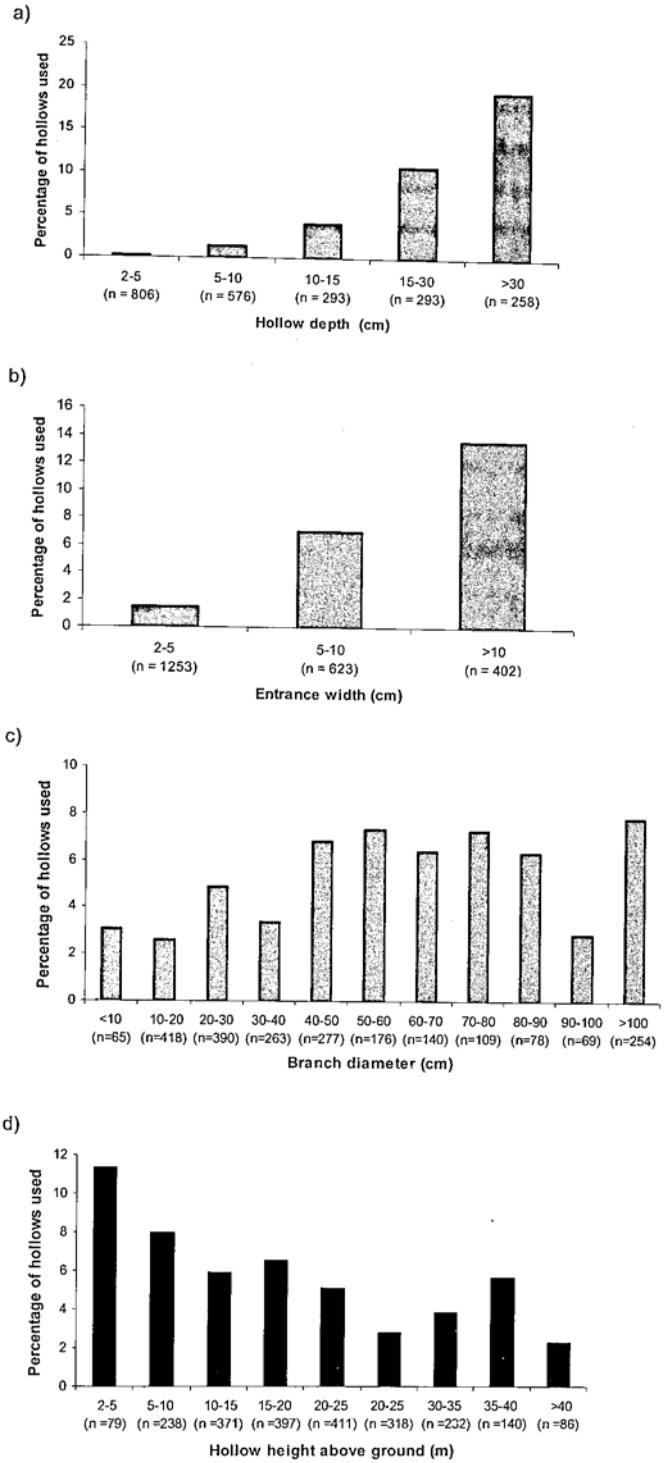


Figure 6.8 The percentage of hollows with evidence of use by fauna in relation to (a) hollow depth, (b) minimum hollow entrance diameter, (c) branch diameter and (d) hollow height above the ground.

Forty-six of the 127 instances of animal use (36.2%) could not be placed in an animal species group. Less than half of the hollows used were detected in the pre-fall survey (Figure 6.9). It was assumed that any evidence collected from a particular hollow indicated use of that hollow by a particular species. However, for the brushtail possum this may not have been the case. On a number of occasions there was evidence of both birds and brushtail possums in a particular hollow. It is therefore believed possible that the hairs were taken into the hollows by the birds. In other situations the hollows were clearly too small for a brushtail possum and so there was either a misidentification of the hair or the hair got there by some other means. The youngest tree to be used by a brushtail possum was thought to be only 87 years old. In this case the hollow was a very open cavity at the fork between two branches. The smallest tree to be used was only 51 cm in diameter, although it was estimated to be 214 years of age and again, the hollow was so small it was unlikely to have been used for denning. Details of the nesting sites where species identification was made are provided in Appendix 3.

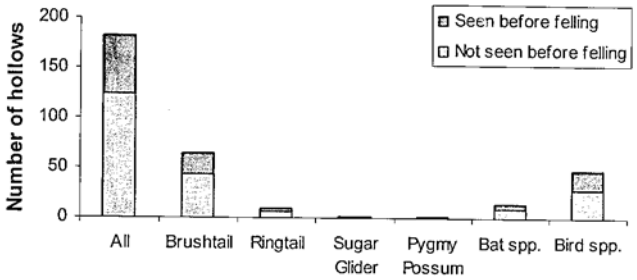


Figure 6.9 The number of hollows that were and were not observed in a standing tree that showed evidence of use by any animal, the brushtail possum, the ringtail possum, the sugar glider, a pygmy possum, a bat or a bird.

Discussion

Use of trees by fauna

The rate at which hollow-bearing trees are used by vertebrate fauna in Tasmania appears to be at the lower end of the range found in studies in mainland Australia (see Table 6.13). In the current study, 22.7% of all trees and 28.6% of hollow-bearing trees 50 cm or greater in diameter showed evidence of use by fauna. The rate of use increased to nearly 50% in trees containing large hollows. In comparison, other areas of Australia report rates of use from 7.5% to 57% (see Table 6.13).

The most comparable study is that of Gibbons (1999) who also looked at *E. obliqua* (among other species) using a method of tree dissection, but in southeastern mainland Australia. Forty-six vertebrate species were believed to use hollows in the study area (Gibbons, 1999), which is greater than the 42 species estimated to use hollows in Tasmania (Chapter 2). The percentage of hollow-bearing trees which showed evidence of use by fauna (57%) found by Gibbons (1999) was higher than was found for the current study (29.2% for similar tree selection and hollow dimensions). The only other work done in Tasmania looking at the rates of tree use was that done by Taylor and Haseler (1993) in dry forest in the northeast of the state. Although similar densities of hollow-bearing trees have been estimated in the area (48 per hectare: Haseler and Taylor unpublished data cited in Taylor, 1991) compared to the current study (12 to 64 per hectare), the rate of use was much lower (0.4 trees per hectare: Taylor and Haseler, 1993) than the 8 to 15 trees used per hectare that was found in the current study. Much of the difference will be due to the fact that Taylor and Haseler (1993) only considered use by four species of bird and much of the faunal evidence found in the current study was of arboreal marsupials. Therefore the current study is expected to provide a more accurate estimate of rates of tree use.

There are two main methods by which the rate of tree retention required has been estimated in Australia. The first method involves summing the product of the density of species and the number of hollows per hectare required by each species (e.g. Smith, 1993; cited by Lamb *et al.*, 1998; Lamb *et al.*, 1998; Smith and Lees, 1998). Some studies then divide this figure by the average number of hollows per tree to reach an estimate of the number of habitat trees required (e.g. Lamb *et al.*, 1998), while other studies assume that sharing of trees does not occur (e.g. Smith and Lees, 1998). Smith and Lees (1998) used this method to estimate that the seven species of arboreal marsupial in southeast Queensland require up to 7.2 trees in dry sclerophyll forest, 5.5 in moist coastal hardwood forest and 8.2 in wet coastal hardwood forest. Smith (1993; cited by Lamb *et al.*, 1998) estimated that 6-13 medium to large hollows and 12-27 small hollows per hectare were required by hollow-users in northeast NSW, with

upper limits applying to high quality sites and lower limits to low quality sites. Lamb *et al.* (1998) suggested this would be provided by 6-13 habitat trees per hectare, if each habitat tree contains, on average, one large hollow and two small hollows. In Queensland, Lamb *et al.* (1998) also used this method but recommended that trees be retained at a third or half the rate at which trees were estimated to be used by hollow-using fauna in order to achieve a compromise between wood production and conservation. The number of trees recommended for retention varied with forest type (six in moist forest, wet forest and dry forest within the range of the greater glider, four in other dry forests areas and two in Cypress forests).

The second main way in which recommended rates of tree retention have been estimated is by comparing the occurrence of species with the density of hollow-bearing trees. Mackowski (1984) found low numbers of arboreal marsupials in northern NSW when there were less than three hollow-bearing trees per hectare. Smith and Murray (2003) concluded that the abundance of arboreal marsupials on the central coast of NSW peak when there are more than 30 hollow-bearing trees per hectare. Although it is unclear how Moloney (2002) reached the recommendation of retaining 5-8 habitat trees per hectare in southeast Queensland, it is presumed that it was based on the fact that the six species of arboreal marsupial considered were only found in areas with at least four hollow-bearing trees per hectare.

Nest boxes and tree-felling surveys are two other methods for establishing the rate of tree use by hollow-using fauna or for making recommendations on rates of tree retention. Rates of nest-box usage in eastern Victoria suggest that up to ten suitable hollows per hectare may be necessary for nine species of animal (see Menkhorst, 1984) while in central Victoria it was concluded that 16 to 24 hollows per hectare are required for the 13 species considered (Calder *et al.,* 1979; cited by Menkhorst, 1984). Gibbons (1999) examined felled trees and estimated that between 7 and 14 trees per hectare showed evidence of use by one of the 46 species of fauna found in southeast Victoria.

In this study it was found that there was a slight increase in the rate at which trees were used in relation to forest wetness. However, the availability of hollows was also shown to increase with forest wetness, as was the general size and age of trees. Many of the hollow-using animals in Tasmania are non-dependent in their use of hollows (Chapter 2) although there may be a number of advantages for those individuals that do use hollows (see Gibbons and Lindenmayer, 2002 pp. 28-29 for a summary). It may therefore be that the apparent increase in hollow use with forest wetness is occurring because of the greater availability of the resource in these areas. Studies which have examined fauna occurrence in relation to forest type have produced contrasting results. Lamb *et al.* (1998) concluded that more hollows are

required by fauna in more productive wet sclerophyll forest than in dry forest. In Victoria, greater densities of brushtail possums were found in dry forest than wet (Bennett *et al.*, 1991). In Tasmania, bird diversity is greater in dry forest areas than wet forest areas, contrasting with mainland studies (Brereton and Taylor, 2000).

One difference between the forest types was that smaller trees could be used by fauna in drier forest. No evidence of use by animals was found in wet forest trees less than 110 cm in diameter. In comparison, some trees less than 70 cm in diameter showed evidence of use in drier forest. However, the proportion of trees in a particular age category that showed evidence of use was similar between the different forest types, with the proportion of trees being used increasing with tree age. This difference in the relationship between tree use, tree diameter and tree age is likely to be explained by the growth rates of trees in these forest types. Trees were found to grow slightly slower in dry *E. obliqua* forest in Tasmania than in either damp or wet forest (Chapter 3). Furthermore, a greater proportion of trees were suppressed in drier forests, resulting in trees that are small in diameter being older than would be expected based on diameter alone. As tree age is related to the onset of hollow production (Chapter 5), these suppressed trees are more likely to have hollows. Alternatively, given that there are more and larger hollows in wet forest than dry (Chapter 5), it may be that the animals have the opportunity to be more selective in their choice of hollows in wet forest, while in the drier forests they are having to settle for the hollows that are available. It is known that in some areas a lack of hollows can limit animal populations, but that in others a decrease in the hollow resource will not affect animal density (Waters *et al.*, 1990; Newton, 1994). Further research is warranted to confirm that the greater use of hollow-bearing trees exhibited in Tasmanian wet forest areas compared to dry was due to a greater availability of tree hollows.

The trees that were being used by fauna were found to have an abundance of hollows. There was also some indication that trees were more likely to be used if they had an abundance of dead wood in the canopy (i.e. more senescent), were large in diameter and older in age, as has been found in other studies (Lindenmayer *et al.*, 1990c; Gibbons *et al.*, 2002). An additional and unexpected variable relating to the use of a tree by fauna was the age of the stand. The results indicated that animals preferred to use hollows in regenerating forest with a mature element. The exact nature of the effect of stand age is difficult to determine due to the difference in stand structure between the forests. All stand ages contained some very large trees, but the mixed age stands had a smaller maximum size (250 cm dbh). Contrary to forest age labels, regrowth with mature forest had a greater proportion of trees examined that were less than 150 years old and 100 cm in diameter than other forest ages (although regrowth forest had a greater proportion that were under 100 years old). Regrowth trees tend

to grow faster and so are younger for the same size (Chapter 3). There are two main reasons which could explain the apparent preference for regrowth forest with a mature element. The first is that animals may prefer this forest type. Younger leaves are more nutritious than older leaves (Monks and Efford, 2006) and the vegetation structure and insect populations can vary with forest age (Baker, 2006). This hypothesis is supported by the results of several studies which have found that arboreal marsupials prefer mixed age forest (Smith *et al.*, 1985; Pausas *et al.*, 1995). Furthermore, in Tasmania brushtail possums are considered a pest in regenerating forest and can occur at higher densities there than in more mature forest (Hocking, 1981). The second is that there is a greater demand for the hollow resource in this forest age. A smaller proportion of trees examined in regenerating forest with a mature element were more than 100 cm in diameter than in any other stand age. This indicates a relatively smaller proportion of trees contained suitable hollows as trees do not begin producing hollows until they reach about 100 cm in diameter (Chapter 5). This may have lead to greater competition for the hollow resource. The reason for increased use in this stand age may also result from a combination of these two causes.

Besides stand age, the only other site level variable that appeared to have some influence was altitude. There was some indication that trees at high altitudes were more likely to be used even when they did not have numerous hollows. Altitude has been related to animal occurrence in other studies, where a decrease in density was found with altitude (Kavanagh and Bamkin, 1995). The trees examined in the current study were shown to be less likely to have a hollow if they were found at high altitude (Chapter 5). Again, while the importance of the relationship was uncertain, it is possible that a smaller proportion of trees contain hollows at high altitudes and so there is greater use of trees which have them. Alternatively, the thermal insulating properties of hollows (Ruczyński and Bogdanowicz, 2005) may mean that non-dependent hollow-using species are more likely to prefer hollows in the higher altitudes. Some evidence was found in the current study to support this hypothesis as the trees used by ringtail possums were found at relatively elevated sites (155 m, 250 m, 387 m, 440 m and 550 m. The mean altitude of sites examined was 270 m ± 134 s.d.: see Appendix 3).

Use of hollows by fauna

The rate at which available hollows were used in the current study was amongst the lowest that has been found in Australia. In the current study, 5.5% of hollows and 24% of large hollows examined showed evidence of use by fauna. In other areas of Australia, between 3.4% and 47% of hollows have shown evidence of occupancy (see Table 6.13). Given that the rate at which trees (as opposed to hollows) were used was within the bounds of what has

been found elsewhere in Australia, these results suggest that the number of hollows being used on a tree is lower than has been found elsewhere.

There are several possible reasons which may explain this discrepancy. Firstly, the low rate of use may in part be due to differences in the definition of a hollow, the trees examined and the technique used in the current study (Table 6.13). It is difficult to compare studies because different tree species form hollows at different rates (Inions *et al.*, 1989) and animals can prefer certain tree species over others (Braithwaite, 1983; Taylor and Haseler, 1993). The most comparable study is that of Gibbons *et al.* (2002) who also dissected *E. obliqua,* among other species in southeastern mainland Australia. However, the rate of hollow use found by Gibbons *et al.* (2002) (43%) was much higher than the comparable figures for the current study (7.1%). A second reason for the low rates of use may be that the number of hollow-using species in Tasmania is at the lower end of many other areas of Australia (Chapter 2). Around 42 vertebrate species are believed to use tree hollows in Tasmania to some extent, of which seven are introduced and/or vagrant species (Chapter 2). This compares with the 46 species which use hollows in southeast mainland Australia (Gibbons, 1999) and 42 species in southwest Western Australia (Abbott and Whitford, 2002). Many of the hollow-using species in Tasmania are migratory bird species which use hollows only for nesting. Furthermore, many of these species have a limited distribution and so the number of hollow-using species occurring in any one area is likely to be even lower. Another explanation for the difference in rates of hollow use in Tasmania compared to mainland Australia is the types of hollows that are available. Of the large hollows examined in the current study, 83.6% were deeper than they were wide but only 46.7% were at least twice as deep as they were wide (unpublished results). Yet the hollows which had evidence of use by fauna (and measurements of entrance width and hollow depth) were on average 5.3 ± 9.7 s.d. times as deep as they were wide (with a median value of 2.5 times as deep). This suggests that the low rate at which hollows are used in Tasmania is because the hollows are too shallow and open to provide adequate shelter.

This theory that hollows are, in general, more shallow in Tasmania than other areas of Australia is supported by the limited literature that is available. For marri trees, (*Corymbia calophylla*) the mean hollow entrance width was 90.2 mm and mean depth was 362 mm (Whitford, 2002). In jarrah trees, (*Eucalyptus marginata*) the mean entrance width was 87.8 mm and depth was 232 mm (Whitford, 2002). This compares with the average of 75 mm entrance width and 160 mm hollow depth for the hollows examined in the present study for which measurement could be taken. Gibbons *et al.* (2000b) examined the dimensions of hollows found in *Eucalyptus obliqua, E. cypellocarpa* and *E. fastigata* in southeastern mainland Australia. Of the hollows that had a minimum entrance width of more than 10 cm,

17% had a minimum depth of 5-10 cm, 24% had a minimum depth of 11-20 cm and 59% had a minimum depth of > 20 cm. If comparable hollows are examined in Tasmania, the rates are 34.7%, 25.6% and 39.7% respectively (unpublished data). This indicates that the average depth of hollows in Tasmania may be substantially smaller than that of hollows on mainland Australia.

The most likely reason for the shallowness of Tasmanian hollows is the relative lack of termites (P. McQuillan pers. comm.). While termites are known to be an important aspect of hollow formation in mainland Australia (Mackowski, 1984; Wormington and Lamb, 1999; Whitford, 2002), Tasmania has very few termite species and most are largely ground-dwelling in nature and/or are found in very low numbers (P. McQuillan pers. comm.). Furthermore, they tend to only attack wood that has been previously decayed. The relative lack of termites in Tasmania most likely means that there are fewer deep hollows suitable for use by fauna. In the present study, 33.7% of hollows showed evidence of invertebrate activity. Only 5.9% of hollows had invertebrate activity as a sole causative agent (of the hollows for which invertebrates were found to contribute to hollow formation, 77.9% were also found to be formed by branch breakage). In comparison, the cause of internal hollowing in southwestern Western Australia was attributed to termites for 19% of hollows (Whitford, 2002). In southeastern mainland Australia, most of the hollows in the main stem were associated with a hollow pipe and termite galleries (Gibbons, 1999). In southeast Queensland, evidence of termites was found in all trees with a diameter greater than 121 cm, 101 cm and 91 cm respectively for the three tree species examined.

The model produced in the current study revealed the importance of hollow depth for fauna. The variables found to be most strongly related to whether a hollow would be used by fauna or not were hollow depth, entrance dimensions and, to a lesser extent, branch diameter and height of the hollow above the ground. Most hollows used were at least 30 cm deep, had an entrance diameter of at least 10 cm, were located on a branch at least 40 cm in diameter and were located close to the ground. It is well established that hollow dimensions are an important variable influencing hollow use (Saunders *et al.*, 1982; Tidemann and Flavel, 1987; Gibbons *et al.*, 2002). Branch diameter is related to hollow size as the maximum size hollow a branch can contain is dictated by its diameter. Larger hollows are also more likely to occur lower on the tree than smaller hollows (Mackowski, 1987). One of the reasons why such large hollows appear to be preferentially selected may be because the majority of evidence identified to species in this study was from the brushtail possum. These animals will probably require hollows at least 5 – 10 cm in entrance diameter (Chapter 2). However, some unusual results were obtained. On occasion, evidence of brushtail possums was found in young trees. It cannot be said definitively if this was a result of use of these trees by

brushtails, or whether hairs were brought there by other animals (although brushtail possums are known to be versatile in their denning requirements: Munks *et al.*, 2004a). However, most other species needed trees to be at least 200 years old before they were used. Exceptions to this were found for a couple of species of birds, but one was some feathers and hairs sitting in the shallow hollow and the other was a nest built into a shallow hollow (fire scar). In comparison to the other species considered, the brushtail possum appeared to be much less selective in terms of the size or age of tree that was used.

Other studies have taken low rates of hollow use to suggest that hollows are not a limiting resource for hollow using fauna (e.g. Edington and Edington, 1972; Mackowski, 1987). Other factors proposed to be limiting have been nesting materials, suitable types of prey and food and interspecific and intraspecific influences such as territoriality (Edington and Edington, 1972; Waters *et al.*, 1990; Recher, 1991). Other research done in Tasmania supports the conclusion reached by the current study that despite the apparent abundance of tree hollows, there may be a shortage in the availability of hollows that are suitable for nesting. The orange-bellied parrot (*Neophema chrysogaster*), the swift parrot (*Lathamus discolor*) and the forty-spotted pardalote (*Pardalotus quadragintus*) are listed as threatened in Tasmania and are examples of species that may be limited to some extent by a lack of suitable nesting sites (Brown and Wilson, 1984; Threatened Species Section, 2006; Webb *et al.*, 2007). Erection of nesting boxes in the southwest of the state for the orange-bellied parrot habitat has had some success, but in this area of believed low hollow-user density (Chapter 2), competition with other species is known to occur (M. Holdsworth pers. comm.). Swift parrots are known to nest communally in an area, meaning high densities of suitable hollows are required to sustain breeding populations, but the location used varies between years according to the food source (Webb *et al.*, 2007). Declines in parrot species in the heavily-modified Midlands area of Tasmania have been attributed to competition for nesting hollows with the introduced common starling (Green, 1983). This situation of observing unused hollows combined with an apparent lack of hollows or competition for nesting sites is not unique to Tasmania. Ambrose (1982) also noted that despite an apparent abundance of hollows in Victoria, competition for nesting sites was frequently observed.

Although hollow dimensions are clearly of great importance to hollow-using fauna, animals are also known to select for other site or hollow attributes (Ambrose, 1982). Of the seven ringtail possums (*Pseudocheirus peregrinus*) sites found in the present study, most were located at sites at higher altitude. The bat species showed some selectivity for east-facing hollows (5 of 9 faced east) and old trees. Bird species also appeared to select north or east facing hollows (of the 28 sites, 10 faced north and 9 faced east) and old trees (Appendix 3). This preference by animals for certain aspects has been previously reported in the literature

(Ambrose, 1982) and supported by anecdotal reports. A forestry worker in the northwest of the state reported that he finds more animals on northern slopes, especially the sections that get the afternoon light. He also stated that sugar gliders are generally found nesting communally in the higher parts of the trees and often in the dead trees or dead sections of the trees.

Recommendations for the selection of habitat trees

Two important questions that need to be answered when planning for the retention of habitat for hollow-using fauna in a particular area are: 'Which trees should be retained?' and 'How many trees should be retained?' The results of the current study can be used to help answer these questions for wet and dry *E. obliqua* forest in Tasmania.

The results showed that the abundance of hollows in a tree, its age, size and degree of senescence were all factors that could be used to predict use of the tree by fauna. Despite the fact that roughly half of the hollows showing evidence of use by fauna were undetected during the pre-fall survey, one of the best ways for predicting faunal use is to count the number of hollows in the standing tree. However, the regression modelling indicated that it is more important to do hollow counts in young and healthy trees than in their more senescent counterparts. However, the poor predictive ability of these models may limit their utility in selecting habitat trees for retention. As hollow abundance and size were some of the most important parameters for predicting tree use, prescriptions for the selection of habitat trees should be at least partially based on the likelihood that a tree will contain many or large hollows.

The most important parameters for estimating hollow abundance were similar to those for predicting tree use; the number of hollows detected, tree age, tree size and senescence (Chapter 5). Trees needed to be more than 100 years old to have a hollow at least 5 cm deep. In order to have a hollow at least 15 cm deep, they had to be 140 years old. The results of the current study have shown that a hollow needs to be greater than 15 cm deep in order to have a high chance of being used by fauna and so it is the large hollow model which should be used to identify suitable habitat trees. The amount of dead wood was related to the likelihood a tree contained at least one large hollow (Chapter 5). However, the best survey model for predicting presence of large hollows was that at least six hollows needed to be seen in the pre-fall survey and the tree diameter needed to be at least 125 cm.

Due to the difference in growth rate between the forest types, it is proposed that in wet forest habitat trees need to be at least 125 cm in diameter and have at least 6 visible hollows. In dry forest areas, smaller trees could be retained, but they should be at least 100 cm in diameter.

A limit of 100 cm was chosen because it has been estimated that trees of this size will be approximately 200 years old (Chapter 3) which is sufficiently old to have a high likelihood of having a large hollow (Chapter 5). Furthermore, only three of the 13 (23%) dry forest sites where a basal area plot was done did not contain a tree of this size. It is likely that examination of a larger area may have found a sufficiently large tree even at these sites. The size of trees to be retained at damp forest sites is dependent on the size of tree available. For those containing many large trees, the prescriptions for selecting trees in wet forests should be used. In the damp forest sites with smaller tree sizes, the prescriptions for dry forest sites should be used.

The results of the current work indicated that between 8 and 15 trees are likely to be used per hectare by fauna, although it should be acknowledged that there is potentially a high level of error in these estimates due to error in the estimates of tree density. These figures were established in two main ways. The first method used the proportion of trees that were found to be used by fauna to estimate rates of tree use. This method gave a higher density of faunal use than was found by the second method, which involved using the predictive models developed in the current study. The higher rates of tree use found by the first method (11.7 ± 3 trees in dry forest, 12.7 ± 4 trees in damp forest and 14.8 ± 7 trees in wet forest) are likely to be the more accurate estimates of hollow use. This is because the models for predicting use of a tree by fauna developed in the current study had poor predictive ability. Trees were used by fauna in all forest types, but the first survey model predicted a high likelihood of occupancy only in regrowth forest that contained a mature element (Figure 6.3a). In the second model, only trees of a large diameter had a high likelihood of being used (Figure 6.3b). While the majority of trees that were used were at least this size, a number of smaller trees in dry and damp forest were also used.

The current rate at which hollow-bearing trees (habitat trees) are retained within logging coupes set by the Forest Practices Code (Forest Practices Board, 2000) is that two to three habitat trees be retained every five hectares (0.4 to 0.6 per hectare). The demand for hollow-bearing trees as has been indicated by this study suggest that current estimates of hollow usage are 20 to 40 times greater than the rate at which trees are being retained within coupes. This strongly suggests that the current tree retention prescriptions are inadequate to maintain current populations of hollow-using fauna in harvested areas.

Table 6.13 A summary of some studies examining the rates of tree use by vertebrate fauna in Australia. "-" indicates it was not specified.

Region	Main tree species	Survey technique	Trees examined	Definition of hollow	Animal species surveyed for	% of trees used	% of hollows used	Reference
Central Highlands, Victoria	*E. regnans, E. delegatensis* and *E. nitens*	Stagwatching	Hollow-bearing trees (>50 cm dbh)	4 cm in entrance diameter (1.5 cm for cracks)	Arboreal marsupials	30.8%	-	Lindenmayer *et al.*, 1990c
Murray River National Park, South Australia	*E. camaldulensis* and *E. largiflorens*	Diurnal surveys and stagwatching	All trees (dbh 36 – 247 cm)	"An opening.... Which could possibly be used"	Birds and mammals	49%	13%	Gates, 1996
Southwest Western Australia	*E. salmonophloia*	Diurnal surveys	Hollow-bearing trees	9 cm entrance diameter and 9 cm deep	Birds (9 species observed)	-	47%	Saunders *et al.*, 1982
East Gippsland, Victoria	Several species including *E. obliqua*	Tree dissection	Hollow-bearing trees	2 cm entrance diameter and 5 cm deep	All species	57%	43%	Gibbons *et al.*, 2002
Southwest Western Australia	*E. marginata* and *Corymbia calophylla*	Tree dissection	A stratified sample of trees according to species, dbh (>20 cm) and crown senescence	2 cm entrance diameter and 10 cm depth	All species	7.5%	3.4% (11.2% of hollows believed to be useable)	Whitford and Williams, 2002
Northeastern NSW	*E. pilularis*	Tree dissection	A stratified sample according to tree dominance and diameter	Unstated	All species	-	11.8%. 1.14 hollows used per tree for 100-119cm dbh 0.51 for 120-139, 2.8 for 140-159 and 2.84 for ≥160	Mackowski, 1987
Northeast Tasmania	*E. obliqua* and *E. amygdalina*	Visual surveys	All trees >10 cm dbh	"Potential entrance to a hollow"	Four species of bird	7.8%	38.1%	Taylor and Haseler, 1993
Central Highlands Victoria	*E. regnans* and *E. delegatensis*	Stagwatching	Trees with hollows located in retained linear strips	Assumed to be as for previous study: 4 cm in entrance diameter	Arboreal marsupials	13.4% but varied greatly between sites	-	Lindenmayer *et al.*, 1993a
Tasmania	*E. obliqua*	Tree dissection	Trees > 50 cm in dbh	2 cm entrance diameter and 2 cm depth	All species	22.7%	5.4%	The current study

Chapter 7
Final discussion

Introduction

Due to the importance of the tree hollow resource for native fauna in Australia, management strategies have been implemented to try and ensure an ongoing supply of hollows. The prescribed rate of habitat tree retention within logging coupes in Tasmanian production forests is among the lowest in the country (Wayne *et al.*, 2006) and the prescriptions for the types of trees to be retained are vague and are based on little empirical data (Forest Practices Board, 2000). The overall aim of this thesis was to assess the tree hollow resource and its use by fauna in two major forest types in Tasmania, with the intention of providing information that could be used to inform the review and improvement of conservation management strategies for hollow-using fauna in Tasmanian production forests. As part of this work, methods involved with studying tree hollows were assessed for their accuracy and utility. To achieve these aims, 391 trees were examined before and after felling in forestry coupes throughout Tasmania. Data was collected on tree growth rings, hollow occurrence and whether the trees were being used by fauna.

A recent publication reviewed most aspects of tree hollow management in Australia (see Gibbons and Lindenmayer, 2002) and so a discussion of the tree hollow issue on a broad scale was considered repetitive and unnecessary. Instead, the main aim of this final chapter is to discuss the management of the tree hollow resource in Tasmania and the application of the current results in developing or improving the conservation management strategy, at a range of spatial and temporal scales, for Tasmanian hollow-using fauna.

The landscape level

The hollow resource is captured to a certain extent in Tasmania's range of formal and informal reserves which are located across the landscape (Figure 2.4). This network of reserved areas provides important habitat for many forest fauna species, including hollow-using fauna. Formal reserves include National Parks, State Reserves, Forest Reserves, Conservation Areas and several other land tenures. However, as discussed in Chapter 2, the distribution of these formal reserves is not evenly distributed, with a large proportion of the reserved area located in the relatively mountainous and unproductive southwest and mid-west region of the state. The largest reservation gap is located in the Midlands (Figure 2.4). Management of the hollow resource is particularly important in these areas of low formal

reservation. Informal reserves, established on both private and state land include streamside reserves, wildlife habitat strips and other areas where harvesting activities are specifically excluded by management zoning. Streamside reserves are strips of native forest retained next to a stream, the width of which depends on the class of the stream (class 1: 40 m; class 2: 30 m; class 3: 20 m) (Forest Practices Board, 2000). Wildlife habitat strips are strips of uncut forest 100 m wide, based on streamside reserves but including links up slopes and across ridges to connect with watercourses in adjoining catchments. It is recommended they be provided every 3-5 km and should connect any large patches of forest which are not to be harvested (Forest Practices Board, 2000).

The majority of habitat important for the maintenance of populations of hollow-dependent fauna, however, is unreserved. Such habitat is found in the extensive areas of unreserved forest on state and private land that is potentially subject to production forestry activities and in the small remnants and isolated trees in agricultural and urban environments. There is generally a lack of regeneration in areas that have been extensively cleared leaving only isolated trees or small remnants (Bennett *et al.*, 1994) which means the future availability of hollows is uncertain. In areas subject to production forestry, rates of regeneration are so high that trees are usually harvested before they reach an age when they will provide hollows for fauna. Careful management is essential in all of these areas outside of the reserve system to ensure that rates of hollow-bearing tree retention and recruitment are sufficient to ensure a perpetual supply of hollows for the maintenance of populations of hollow-dependent fauna across the landscape and into the future.

Forest type

Forests have been classified into different communities based on the understorey vegetation (Ashton, 1981; Forest Practices Authority, 2005). Wet sclerophyll forests are characterised by having broad-leaved species, ferns or a rainforest element dominant in the understorey, while dry sclerophyll forests have narrow-leaved, grassy or mixed understorey dominance (Ashton, 1981; Kirkpatrick *et al.*, 1988; Forest Practices Authority, 2005). Different forest communities will vary in fire regimes, soil nutrition, rainfall, and even tree size and age. It is therefore likely that a difference exists in both the use of trees by fauna and the availability of hollows between these forest types.

An examination of the hollow forming processes showed some differences between the forest types; fire contributed more to hollow formation in dry forest than wet forest and invertebrates contributed more in wet forest than dry.

Fire contributes to the hollow forming process by enlarging pre-existing hollows, creating scars which can then be exposed to further hollow-forming processes (Inions *et al.*, 1989) and by placing the trees under physiological stress which causes a greater number of dead branches. Hollows are more likely to occur on dead branches (Gibbons *et al.*, 2000a). In Western Australia fire was shown to reduce the time taken for trees to become hollow-bearing by about 100 years (Inions *et al.*, 1989). This would suggest that dry forest should have more hollows than wet forest. However, it has also been noted that to promote hollow formation a fire will, in most cases, need to be of high intensity (Adkins, 2006). Therefore, the low intensity of fires generally experienced in dry forests means that scars are produced, which can contribute to counts of hollow abundance, but they will rarely form large hollows.

The incidence of termites in Tasmanian eucalypts increases with tree diameter (Elliott and Bashford, 1984), and it is possible that it is the size structure of wet forests compared to dry that results in the greater relative importance of termites in wet forest areas. Approximately one third of the hollows examined showed some signs of damage by invertebrates, although most of these also showed evidence of other important hollow-forming processes, such as branch breakage (Chapter 5). No differentiation was made between damage by termites or damage by other wood-boring invertebrates such as beetles (Hopkins *et al.*, 2006). In other areas of Australia, termites are considered a major factor in the formation of hollows (Mackowski, 1984). However, relatively few species of termites occur in Tasmania compared to mainland Australia and those that do are either found in low numbers or are largely ground-dwelling. Furthermore, most of these species only colonise damp or dead wood (P. McQuillan pers. comm.). It was proposed in Chapter 6 that this relative lack of termites, one of the major hollow-forming processes in Australia, results in a relative lack of deep hollows in Tasmania compared with mainland Australia.

The most important process contributing to hollow formation identified in the present study was branch breakage. There are many theories as to why eucalypts continually shed branches. These have been summarised in Gibbons and Lindenmayer (2002 p. 40) and include competition, structural limitations and physical damage including the influence of fire and wind. No difference between forest types was found in the propensity of trees to shed branches despite competition and structural damage being likely to vary between the different forest types. Therefore, *E. obliqua* in Tasmania may have structural and/or physiological limits that are the major influence on branch breakage and, therefore, hollow formation. This theory is consistent with the observation that the trees examined indicated an age at which hollow formation began to occur, with the abundance of hollows in trees older than this limit being more strongly related to the amount of dead wood in the canopy than tree age (Chapter 5).

Despite the difference in the occurrence of fire and invertebrates between the different forest types, no difference was found in the rate of hollow formation, possibly due to the similar effect of branch breakage between the forest types (Chapter 5). However, the proportion of trees within a particular size class that contained hollows decreased with forest wetness (Chapter 5) as a result of the difference in growth rates between the forest types (Chapter 3). Yet in both Chapters 5 and 6 it was shown that the density of hollows was greater in wet forest areas than dry. Given the similar rate of hollow formation between the different forest types, it may be concluded that the difference in hollow density that occurs between the wet and dry forests is a result of the greater size and age of trees that were generally found in wet forest areas (Chapter 6).

Although trees grow faster in wetter forest (Chapter 3) and the maximum age of trees is possibly less in dry forest than wet (see Banks, 1997), the difference in the stand structure of the different forest types examined in the current study is largely a result of the different disturbance histories. The main disturbances experienced by Tasmanian forests are fire, clearing and timber harvesting. All of these disturbances have occurred with greater frequency in drier forest (Bosworth *et al.*, 1976; Bowman and Kirkpatrick, 1984). However, once land has been cleared it generally remains cleared, so the current stand structure found in Tasmanian forests is largely a result of disturbance by fire and timber harvesting. Fires occur more frequently in dry forest than wet, but at a lower intensity which generally does not cause the death of many live trees (see Gibbons and Lindenmayer, 2002 p. 117-120). Therefore, the reduced size and age of trees found in dry forest compared to wet forest in Tasmania is unlikely to be due to the different fire regimes. It therefore appears that the difference in stand structure between Tasmanian wet and dry forests is largely due to differences in the history of timber harvesting for each forest type.

Timber harvesting has occurred in Tasmania since European settlement in 1803 (Hickey and Wilkinson, 1999) and has been mostly concentrated in the more easily accessible dry forest in the eastern half of the state (Bosworth *et al.*, 1976; Coulson and Coulsen, 1981). Initially timber harvesting involved selective cutting with little deliberate effort to ensure regeneration (Hickey and Wilkinson, 1999). Harvesting began occurring with greater frequency in the wetter forests in the 1960s using a clearfelling technique. Clearfelling was then introduced in the drier forest in the 1970s. This was, however, short lived and partial harvesting was then introduced to dry forest in the 1980s (Hickey and Wilkinson, 1999). A number of the study sites examined in the present study that were located in wet forest had no recorded history of logging. In contrast, most dry forest sites examined showed evidence of prior logging (Chapter 5). Given that the difference in hollow occurrence between the

different forest types was largely due to the age structure of the forests, it is almost certain that the availability of hollows has been greatly reduced in Tasmanian dry forests as a result of the timber harvesting that has occurred there over time.

In the present study, a slight increase in the rate at which trees are used by fauna with forest wetness was observed (Chapter 6). However, the number of hollow-using species is possibly greater in dry forest than wet forest (Chapter 2). The limited data available suggests there is little difference between wet and dry forests in the relative densities of a hollow-using fauna. Mean bat activity was found to be similar between mature wet forest and mature dry forest with an *Acacia* understorey but lower in mature dry forest with a *Banksia* understorey (Duncan, 1995). Low capture rates of pygmy possums make comparisons difficult, but similar numbers of little pygmy possums were caught in wet and dry forest while four eastern pygmy possums were caught in wet forest and none in dry (Duncan, 1995). Similar trapping success of brushtail possums was found in wet and dry forest (Fitzgerald, 1984). Bird surveys in the Mount Wellington range showed little consistent difference in the densities of hollow-using bird species between wet sclerophyll forest and dry (Ratkowsky and Ratkowsky, 1977). These results suggest that the greater use of trees in wetter forest may be an artefact of the increased availability of hollows of all sizes in wet forest rather than being the result of a greater demand for hollows in these areas.

The local level

Rates of tree use

In Chapter 6 it was found that 28.6% of the hollow-bearing trees examined and almost 50% of trees with large hollows showed evidence of use (Chapter 6). This rate of use was at the lower end of the scale found in other areas of Australia (Gates, 1996; Gibbons *et al.*, 2002; but see Whitford and Williams, 2002). It was estimated that between 8 and 15 trees per hectare were used. This is markedly greater than the rate at which trees are currently being retained within logging coupes in production forests in Tasmania (0.4 to 0.6 per hectare). However, there are a number of factors that need to be considered before a decision is made on the number of trees that need to be retained for faunal use in production forest areas.

It was also shown that a number of the hollow-using fauna in Tasmania are not strictly dependent on hollows, but can also use other nesting locations (Chapter 2). The five possums found in Tasmania use hollows for denning whether with young or not, but none of these species are totally reliant on hollows and all can use other locations. All of the eight bat species can use a variety of roosting sites, although large tree hollows are thought to be the most desired location for breeding (R. Brereton pers. comm.; Taylor and Savva, 1988). Only

one of the 29 bird species examined, the owlet nightjar (*Aegotheles cristatus*), was considered to be dependent on tree hollows for both breeding and roosting. Nineteen species of bird were considered to be dependent on tree hollows for nesting but not roosting. Five bird species do use tree hollows for nesting but can also successfully use other sites. Four further bird species considered were thought to rarely use tree hollows (Chapter 2). Although the consequences of using alternative nesting sites is unknown (e.g. impact on fecundity), these figures indicate that the maximum rate at which hollows are used is not necessarily the rate at which hollows need to be maintained in order to sustain current populations of hollow-using species in Tasmania.

Although it is possible that some hollows which were being used were overlooked due to evidence not being found, the estimated rate of use (8 to 15 trees per hectare: see Chapter 6) may be greater than the rate at which trees need to be retained for several reasons. Firstly, the density of large hollow-bearing trees in the landscape may have been overestimated due to the small plot sized used to sample stand structure (0.25 ha, see Chapter 6). This may have resulted in an overestimate for the rate of tree use. Secondly, it was found that some of the hollows displaying evidence of use by fauna were of a dimension that indicated it was unlikely that they were being used for breeding. Use was considered to be indicated if even one hair or feather was found. The possibility of an overestimation is supported by the fact that a similar study to the current one done in forest containing *E. obliqua* in East Gippsland, Victoria (Gibbons, 1999), estimated that between 7 and 14 hollow-bearing trees per hectare were used by fauna. A slightly greater number of hollow-using species occur in East Gippsland (46) compared to Tasmania (42, Chapter 2). Unless Tasmanian hollow-using fauna occur at greater densities than mainland species (a lack of information on the densities of Tasmanian species precludes a comparison), it is expected that the rate at which hollow-bearing trees are used will be related to the number of species using them. Therefore the rate of use of 8 to 15 trees per hectare estimated in the current study is likely to be an overestimate.

However, it is likely that high rates of use will be observed in some areas. For example, swift parrots (*Lathamus dicolor*) are a threatened species which nest communally (Webb *et al.*, 2007). Therefore a large number of suitable hollows will be required in the areas of forest in which this species nests. It therefore appears that the number of trees to be retained needs to be a flexible parameter, varying with the availability of hollows and the requirements of the species that occur in the area. This flexibility has been recommended in other areas of Australia. For example, in areas where the threatened Leadbeater's possum (*Gymnobelideus leadbeateri*) is found, it has been recommended that trees be retained at different rates according to the importance of the area for this species (Smith *et al.*, 1985). It is

recommended that a similar system be implemented in Tasmania with the rate of tree retention reflecting both the conservation status of the species present in the area and the availability of hollows in the landscape.

Harvesting regimes

The tree retention measures currently prescribed in the Tasmanian Forest Practices Code (Forest Practices Board, 2000) vary according to the harvesting regime used, which in turn varies according to forest type. Dry forest areas are generally harvested using a variety of partial harvesting methods (Table 7.1) (Forestry Commission, 1994), which allows retention of hollow-bearing trees in clumps within a coupe. Harvesting in wet forest areas generally involves clear-fell, burn and sow methods (Table 7.1) (Forestry Commission, 1994, p. 68) and the burning of these areas makes the retention of clumps of hollow-bearing trees within the harvesting unit problematic. "In coupes where high intensity burning is required to achieve regeneration or where cable harvesting is used (mainly clearfell coupes), wildlife habitat clumps should be retained along the boundary of the coupe where they can be protected from disturbance" (Forest Practices Board, 2000). The retained clumps are recommended to be about one hectare in size (50 by 200 m) and to be retained every 200 m along the coupe boundary in areas further than 200 m from other reserved areas. The degree to which this technique captures suitable habitat for fauna has not been determined but, given the prescriptions, it will be influenced by the size of the area harvested.

Partial harvesting systems implemented in dry forest areas in Tasmania might be expected to retain more hollows for fauna than clearfelling systems implemented in wet forest areas. However, the amount and type of habitat that is likely to be provided by the partial harvesting system depends on the specific method being implemented, with the majority of techniques providing little additional habitat or, where some habitat is provided, only for a brief period. For example, although retention of advanced regeneration and potential sawlogs provides many ecological benefits, such as preventing erosion and maintaining a food source for some species, it is unlikely to provide any habitat for hollow-using fauna. Some harvesting systems may provide some faunal habitat as larger trees may be retained. Shelterwood involves retaining a proportion of the stand to provide shelter for the regenerating plants and seed tree retention involves retaining scattered large trees as a source of seeds. However, the trees retained using these harvesting systems are generally felled about 10 years after logging, which is well before the regenerating trees will contain hollows (Forestry Commission, 1994). Selective logging is generally applied to open, multi-aged stands with an adequate stocking of sawlogs. Only selected merchantable trees are logged, retaining all other growth. A high proportion of the mature stems retained should be potentially merchantable, since the retention of non-merchantable trees will lower the future

productivity of the stand (Forestry Commission, 1994). Thus, trees which have the greatest likelihood of providing good faunal habitat are unlikely to be retained. The practice of selective harvesting is rarely implemented (Forestry Commission, 1994, p. 69). It can therefore be stated that despite the numerous other benefits that are provided by the partial harvesting systems currently implemented in dry forests in Tasmania, few tree hollows are expected to be retained using these methods in the long term and so equal effort is required in selecting appropriate trees for retention in both wet and dry forest types.

Table 7.1 A summary of the different harvesting techniques and where they are generally implemented. [a]

Type of forest required	Reforestation technique	Where most appropriate
Even-aged forests	Clearfell, burn and seeding by land or air	Used in both wet and dry forests, but particularly in even-aged wet forests which have a dense understorey shrub layer.
	Clearfell and use of lignotuber, coppice, advanced growth or seed from felled trees	Generally in drier forests.
	Seed tree retention	Where retained trees (7-15 per hectare) carry a suitable seed crop and an appropriate seed bed is available or can be prepared. This technique is appropriate in lower quality dry forest. Burning or scarification may be needed to create a seed bed.
	Shelterwood	Mostly implemented at high altitudes, exposed sites. The shelterwood trees are generally felled after approximately 10 years when regeneration has occurred.
	Aggregated retention	Technique currently being investigated for old-growth wet forest areas as an alternative to clearfelling. Involves the retention of large tree clumps in a mosaic pattern.
Uneven-aged forests	Advanced growth retention	Sites carrying an adequate stocking of vigorous advanced regrowth up to potential sawlog size.
	Potential sawlog retention	Sites with advanced growth currently marketable as pulpwood but which has the potential to grow into higher value products.
	Single tree or group selection	Drier forest types where markets or non wood values limit a higher level of utilisation. Disturbance from harvesting or cool burning is used to provide a seed bed.

[a] Reproduced from the Forest Practices Code (Forest Practices Board, 2000).

How well are management prescriptions for the local level met?

In order to fully assess the adequacy of tree retention prescriptions in logging coupes in Tasmanian forests, it is not sufficient to only ensure the prescriptions provided are sound. It is also necessary to establish the degree to which the prescriptions are actually being adhered

to. Two studies have assessed the degree to which current tree-retention prescriptions are being applied in dry forests in Tasmania, with contrasting results.

The degree to which habitat clump retention meets the current guidelines was assessed by Duhig *et al.* (2000) who found that, in general, the current area and tree requirements are being met. The average size of clump found was 0.09 ha ± 0.05 (n = 29) and the majority of clumps (96%) contained 2-3 'habitat trees' and a number of regrowth trees. In their study, a 'habitat tree' was defined as a tree of a particular form that was at least medium to large in size and had at least one visible hollow. However, in another study that assessed the implementation of swift parrot foraging habitat retention measures within logging coupes, the management prescriptions were poorly met (Munks *et al.*, 2004). Although this study did not focus on the retention of tree hollows, it does give some insight into how well tree selection and retention prescriptions are applied in Tasmania. It was found that only 16% of the retained clumps of trees contained 2-3 mature *Eucalyptus globulus* as per the prescription. The results suggested that the selection of clumps did not consistently target the trees required, although this was potentially due to the structure of the forest and the inadequacy of the prescriptions. It is therefore concluded that the adequacy of the selection of habitat clumps is variable, which emphasizes the importance of active selection of appropriate areas.

In wet forest areas, wildlife habitat clumps are generally not implemented throughout the coupe, instead being retained in patches around the coupe boundary (Forest Practices Board, 2000; Wayne *et al.*, 2006). The degree to which the current prescriptions are currently being met in wet forest is unknown. The models developed during the present study will allow an assessment of the amount of habitat that is being captured as a result of current forest management.

Future tree retention

The rate of mortality for retained trees is generally higher than that of trees within a forest stand due to windthrow and damage sustained during the logging process (Lindenmayer *et al.*, 1997a; Duhig *et al.*, 2000; Gibbons *et al.*, 2000b; but see Whitford and Williams, 2001). Furthermore, trees age, senesce and eventually die. "Failure to replace hollow-bearing trees as they are lost will result in prolonged temporal gaps in the resource that will not only reduce the area of suitable habitat for hollow-using fauna, but could also fragment populations of species unable to occupy areas lacking hollows" (Gibbons and Lindenmayer, 2002, p. 116). It is therefore important when managing an area for tree hollows to consider not just the current availability of hollows, but also the future availability for more than one rotation of harvesting.

Several studies have established that in order to have a perpetual supply of hollows, new 'recruitment' trees need to be retained for the future hollow resource they will provide (Lindenmayer *et al.*, 1990a; Ball *et al.*, 1999). In order to assess the availability of recruitment trees, rates of tree mortality due to windthrow during harvesting, rates of natural mortality due to senescence and rates of hollow formation need to be established. The average rotation length for native eucalypt forest in Tasmania is 90 years for extensively managed forest and 65 years for thinned regrowth forest (Whiteley, 1999). Other areas of Australia specify the number of recruitment trees that need to be retained in order to perpetuate the hollow resource (Wayne *et al.*, 2006). Although it is assumed that recruitment trees will be retained by default in habitat clumps in Tasmania, no specific prescriptions are provided on the number or type of trees to be retained. Furthermore, no attention has been given to the retention protocols that are required upon successive harvests. For example, should the same trees be retained or not? How many new trees should be retained at each successive harvest? Should the newly retained trees be added on to the established clump or should they be established as a new clump elsewhere on the coupe? In order to address the issue of future tree retention, some modelling of tree and hollow recruitment and senescence needs to be done. This thesis provides some of the information required for such modelling.

The use of retained trees

Animals are selective in their choice of den sites (Saunders *et al.*, 1982; Inions, 1985; Haseler and Taylor, 1993; Bunnell *et al.*, 2002). In Chapter 6 it was found that only half of the trees containing large hollows showed evidence of being used by fauna. This potentially indicates that only half of the habitat trees retained are likely to be used. However, it was also found that rates of tree use are higher in regenerating forest with a mature element. Consequently, higher rates of use may be found in habitat clumps than would ordinarily be expected. The degree to which the retained habitat clumps are used needs to be established. But the question of utility is further clouded by issues of competition between and within species for the hollow resource (Ambrose, 1982; Green, 1983), the degree to which co-occupancy of a tree occurs (Lindenmayer *et al.*, 1997c) and the importance of the spatial arrangement of hollows (Lindenmayer *et al.*, 1990; Gibbons and Lindenmayer, 1997a). It is therefore apparent that further study is required to determine the degree to which the habitat clumps are actually being used.

A small number of studies have been done in Tasmania on the effectiveness of retaining trees in production forest areas. Bird surveys in dry sclerophyll forest before and after logging indicated a lower bird species richness and abundance in retained corridors (MacDonald *et al.*, 2005). Although some logging sensitive birds including the hollow-

nesting green rosella (*Platycercus caledonicus*) were found in corridors after logging, others such as the dusky robin (*Melanodryas vittata*) were not. Such differences, however, may lessen as the surrounding vegetation matures. Wapstra and Taylor (1998) also looked at retained trees in harvested dry forest in Tasmania. They examined trees which were known to previously be used by hollow-nesting birds. They found that there was a higher incidence of reuse of nest trees in the retained strip and the logged area compared with the unlogged area. These results compare to those of Chapter 6 that use of trees was higher in regenerating forest with a mature element. The results indicate that retained trees are likely to be used by fauna.

When it has been established that retained trees are being used by fauna, it then needs to be established if the tree retention prescriptions currently implemented are adequate to cater for the diverse requirements of hollow-using fauna in Tasmania. Although it was acknowledged in Chapter 2 that the rates of tree use are likely to be lower in Tasmania than other forested areas of Australia, it was shown in Chapter 6 that the current retention of at least two habitat trees per five hectares in logging coupes is unlikely to meet the requirements of animal populations in the local area. The results of this thesis provide a strong framework for research into the rates at which retained trees in production forests are used.

The tree level

The use of hollows by vertebrate fauna appeared to be lower in Tasmania than in other areas of Australia, with only 5% of hollows showing evidence of use (Chapter 6). Hollow dimensions were the variables most strongly related to whether a hollow was used by fauna or not. The low rate of hollow use in Tasmania appears to be due to a lack of deep hollows, which is probably the result of the depauperate termite population in Tasmania (Chapter 6). This result indicates that even though a tree contains a hollow with a large entrance, it is not necessarily suitable for use by fauna. This confirms the necessity of carefully selecting trees to be retained. It also indicates that even if trees are carefully selected for retention, a number are still likely to be unsuitable for use by fauna.

The current Tasmanian guidelines on habitat tree selection within logging coupes are "on the basis of size and the presence of hollows or the potential to develop hollows over time" (Forest Practices Board, 2000). These prescriptions are imprecise and therefore open to interpretation. Habitat tree and clump selection within a particular harvesting unit is undertaken by a Forest Practices Officer, although this is not always the case (Duhig *et al.*, 2000). The provision of clear and specific criteria is required to ensure appropriate trees are selected for retention by untrained personnel and would also assist in the training of Forest

Practices Officers. Furthermore, as outlined in Chapter 4, it can be very difficult to locate hollows in standing trees meaning that, even for trained officers, the selection of the best habitat trees may be difficult. Guidelines which are clear and concise are more likely to be adhered to and are therefore more likely to be effective.

It has been recommended by some authors that trees retained for habitat should target the larger hollows, as they are rarer in the landscape and trees with large hollows will generally also have some smaller hollows (Abbott, 1998; Whitford, 2002). This recommendation was supported by the results of Chapter 6 which were that trees are unlikely to be used by hollow-using fauna unless they contain at least one large hollow. In Tasmania, three bird species are restricted to using large hollows, it is likely that the bats are reliant on large hollows for roosting colonies and large hollows are also frequently used by a number of other species (Chapter 2). Therefore, a recommendation that a habitat tree should contain at least one large hollow is appropriate.

As has been found in studies elsewhere in Australia (Lindenmayer *et al.*, 1993; Gibbons *et al.*, 2000a; Whitford, 2002; Harper *et al.*, 2004), hollow occurrence in Tasmanian forests was related to tree age, size, senescence and the number of visible hollows (Chapter 5). The models indicate that when trees reach about 50 cm in diameter, or 75 years of age, the first trees begin producing hollows. However, the rates of hollow occurrence remain low until a threshold of about 100 years or 80 cm is reached. After about 140 years, or 100 cm in diameter, it is more common to find slightly deeper hollows, with almost 60% of trees examined having hollows which are at least 10 cm in entrance diameter and 15 cm deep. These models were a summary of all trees with no differentiation being made between forest types, although it was found that trees grew slightly slower in dry forest than wet. The best general model found for predicting the occurrence of large hollows was that more than six hollows should be visible in the tree and the tree should be over 124 cm in diameter.

However, it is not sufficient to merely retain trees with hollows. Animals are known to be selective in their choice of habitat (Braithwaite *et al.*, 1984; Smith and Lindenmayer, 1992; Kavanagh, 2000; Alexander *et al.*, 2002). It is therefore important to establish the characteristics of trees most likely to be used by fauna in order to facilitate retention of suitable trees. The variables most strongly related to the use of a tree were again, hollow abundance, tree size and senescence, which again, was consistent with previous work (Taylor and Savva, 1988; Lindenmayer *et al.*, 1996; Gibbons *et al.*, 2002; Wayne, 2005). Trees were also more likely to be used in areas of regenerating forest with a mature element. However, the models predicting the use of a tree by vertebrate fauna had poor predictive ability. As the abundance of large hollows was found to be related to tree use, it was

recommended that the selection of habitat trees should be based on the presence of large hollows.

As the rate of growth of trees was found to vary between the different forest types (Chapter 3), different prescriptions for the types of trees to be retained are required for wet and dry forests in Tasmania. It is recommended that habitat trees in wet forest areas should be at least 125 cm in diameter and have at least six visible hollows. In dry forest areas a tree should be at least 100 cm in diameter and have at least six visible hollows. Where these prescriptions cannot be met, the next largest tree should be retained.

Conclusion

Approximately 54.4% of the forest in Tasmania is unreserved and potentially available for forestry activities. The hollow resource in 'off-reserve' areas needs to be managed at the landscape, local and tree level in order to ensure a perpetual supply of tree hollows is available for native fauna. Issues that need to be addressed at the landscape level are the degree to which the current reserve system adequately captures the hollow resource in the different forest communities and whether the spatial distribution of reserves is adequate. Attention also needs to be paid to the ongoing retention of remnant hollow-bearing trees in agricultural and urban areas and the degree to which hollow availability is limiting populations of native fauna. At the local level, attention needs to be paid to the number and spatial distribution of harvesting units and the location of retained areas within these units. At the tree level we need to ensure that suitable habitat trees and recruitment trees are being retained for fauna. This thesis has improved substantially the ecological understanding of the abundance and distribution of tree hollows and their use by vertebrate fauna in Tasmania's forests and has helped to address some of these issues.

References

Abbott, I., 1998. Conservation of the forest red-tailed black cockatoo, a hollow-dependent species in the eucalypt forests of Western Australia. Forest Ecology and Management 109, 175-185.

Abbott, I., Loneragan, O., 1983. Growth rates of jarrah (*Eucalyptus marginata*) in relation to site quality in cut-over forest, W.A. Australian Forestry 46, 91-102.

Abbott, I., Whitford, K., 2002. Conservation of vertebrate fauna using hollows in forests of south-west Western Australia: strategic risk assessment in relation to ecology, policy, planning, and operations management. Pacific Conservation Biology 7, 240-255.

Adkins, M.F., 2006. A burning issue: using fire to accelerate tree hollow formation in *Eucalyptus* species. Australian Forestry 69, 107-113.

Alcorn, P.J., Dingle, J.K., Hickey, J.E., 2001. Age and stand structure in a multi-aged wet eucalypt forest at the Warra silvicultural systems trial. Tasforests 13, 245-259.

Alexander, J.S.A., Scotts, D.J., Loyn, R.H., 2002. Impacts of timber harvesting on mammals, reptiles and nocturnal birds in native hardwood forests of East Gippsland, Victoria: a retrospective approach. Australian Forestry 65, 182-210.

Allen, S.E., 1974. Chemical analysis of ecological materials. Blackwell Scientific Publications, Oxford.

Ambrose, G.J., 1982. An ecological and behavioural study of vertebrates using hollows in eucalypt branches. Ph.D. Thesis, La Trobe University, Bundoora, Vic., Australia.

Anonymous, 1919. Order Psittaciformes, Family Loriidae, Genus *Trichoglossus*. South Australian Ornithologist 4, 38-39.

ANZECC and MCFFA (1997). Nationally agreed criteria for the establishment of a comprehensive, adequate and representative reserve system for forests in Australia. A report by the joint ANZECC / MCFFA National forest policy statement implementation sub-committee. Canberra, ACT.

Ashton, D.H., 1981. Fire in tall open-forests (wet sclerophyll forests). In: Gill, A.M., Groves, R.H., Noble, I.R. (Eds.), Fire and the Australian Biota. Australian Academy of Science, Canberra, pp. 339-366.

Australasian Bat Society Inc., 2005. Bats and wind farms. Brochure by the Australasian Bat Society Inc., Tamarama NSW.

Australian National Wildlife Collection, RGH Green collection. CSIRO, Canberra.

Bai, M.L., Wichmann, F., Muhlenberg, M., 2003. The abundance of tree holes and their utilization by hole-nesting birds in a primeval boreal forest of Mongolia. Acta Ornithologica 38, 95-102.

Baker, S.C., 2006. A comparison of litter beetle assemblages (Coleoptera) in mature and recently clearfelled *Eucalyptus obliqua* forest. Australian Journal of Entomology 45, 130-136.

Ball, I.R., Lindenmayer, D.B., Possingham, H.P., 1999. A tree hollow dynamics simulation model. Forest Ecology and Management 123, 179-194.

Banks, J.C.G., 1982. The use of dendrochronology in the interpretation of the dynamics of the snow gum forest. Ph.D. Thesis, Australian National University, Canberra, Australia.

Banks, J.C.G., 1990. The fire history for two forest types in Glenboag State forest, NSW, Joint Scientific Committee on the South-East Forests, Canberra.

Banks, J.C.G., 1993. Tree-ring analysis of two Mountain Ash trees *Eucalyptus regnans* F. Muell. from the Watts and O'Shannassy Catchments, Central Highlands, Victoria. Department of Forestry, School of Resource and Environmental Management, Canberra.

Banks, J.C.G., 1997. Tree ages and ageing in yellow box. In: Dargavel, J. (Ed.) Australia's ever-changing forests III. Centre for Resource and Environmental Studies, Australian National University, Canberra, pp. 35-47.

Barrett, G., Silcocks, A., Cunningham, R., 2002. Australian Bird Atlas (1998-2001) Supplementary Report No.1 - Comparison of Atlas 1 (1977-1981) and Atlas 2 (1998-2001). Birds Australia, Melbourne.

Barrett, G., Silcocks, A., Barry, S.C., Cunningham, R., Poulter, R., 2003. The new atlas of Australian birds. Royal Australasian Ornithologists Union, Melbourne.

Bekessy, S., Fox, J., Munks, S., Wintle, B., 2004. PVA for Ringtail Possum (*Pseudocheirus peregrinus convolutor*). In J.C. Fox, T.J.Regan, S.A. Bekessy, B.A. Wintle, M.J. Brown, J.M. Meggs, K. Bonham, R. Mesibov, M.A. McCarthy, S.A. Munks, P.

Wells, R. Brereton, K. Graham, J. Hickey, P. Turner, M. Jones, W.E. Brown., N. Mooney, S. Grove, K, Yamada and M.A. Burgman. Linking landscape ecology and management to population viability analysis. Report 2: Population viability analysis for eleven forest dependant species. A project by the University of Melbourne prepared for Forestry Tasmania.

Bell, H.L., 1979. Tree martins nesting in buildings. Australian bird watcher 8, 102.

Bell, P., Mooney, N., Wiersma, J., 1997. Predicting essential habitat for forest owls in Tasmania, Report to the Tasmanian RFA Environment and Heritage Technical Committee, Hobart.

Bell, P.J., Mooney, N., 2002. Distribution, habitat and abundance of masked owls (*Tyto novaehollandiae*) in Tasmania. In: Newton, I., Kavanagh, R., Olsen, J., Taylor, I. (Eds.), Ecology and conservation of owls. CSIRO Publishing, Collingwood, Victoria.

Bennett, A.F., Lumsden, L.F., Nicholls, A.O., 1994. Tree hollows as a resource for wildlife in remnant woodlands: spatial and temporal patterns across the northern plains of Victoria, Australia. Pacific Conservation Biology 1, 222-235.

Bennett, A.F., Lumsden, L.F., Alexander, J.S.A., Duncan, P.E., Johnson, P.G., Robertson, P., Silveira, C.E., 1991. Habitat use by arboreal mammals along an environmental gradient in north-eastern Victoria. Wildlife Research 18, 125-146.

Berg, A.B., Ehnstrom, B., Gustaffson, L., Hallingback, T., Jonsell, M., Weslien, J., 1994. Threatened plant, animal and fungus species in Swedish forests: distribution and habitat associations. Conservation Biology 8, 718-731.

Blakers, M., Davies, S.J.J.F., Reilly, P.N., 1984. The atlas of Australian birds. Melbourne University Press, Melbourne.

Bosworth, P., Dorney, N., Tarran, A., 1976. Application of island biogeographic principles to the selection and management of Tasmanian dry sclerophyll reserves. Masters Thesis, University of Tasmania, Hobart.

Bowman, D.M.J.S., Kirkpatrick, J.B., 1984. Geographic variation in the demographic structure of stands of *Eucalyptus delegatensis* R. T. Baker on dolerite in Tasmania. Journal of Biogeography 11, 427-437.

Bradshaw, F.J., Rayner, M.E., 1997a. Age structure of the karri forest. 1. Defining and mapping structural development stages. Australian Forestry 60, 178-187.

Bradshaw, F.J., Rayner, M.E., 1997b. Age structure of the karri forest: 2. Projections of future forest structure and implications for management. Australian Forestry 60, 188-195.

Braithwaite, L.W., 1983. Studies on the arboreal marsupial fauna of eucalypt forests being harvested for wood pulp at Eden, New South Wales. 1. The species distribution of animals. Australian Wildlife Research 10, 219-229.

Braithwaite, L.W., Dudzinski, M.L., Turner, J., 1983. Studies on the arboreal marsupial fauna of eucalypt forests being harvested for woodpulp at Eden, N.S.W. 2. Relationships between the fauna density, richness and diversity, and measured variables of the habitat. Australian Wildlife Research 10, 231-247.

Braithwaite, L.W., Turner, J., Kelly, J., 1984. Studies on the arboreal marsupial fauna of eucalypt forests being harvested for woodpulp at Eden, N.S.W. 3. Relationships between faunal densities, eucalypt occurrence and foliage nutrients, and soil parent materials. Australian Wildlife Research 11, 41-48.

Brereton, 1997a. Management prescriptions for the swift parrot in production forests. Report to the Tasmanian RFA Environment and Heritage Technical Committee, Hobart.

Brereton, R., 1997b. Management prescriptions for the swift parrot in production forests. Report to the Tasmanian RFA Environment and heritage Technical Committee, Hobart.

Brereton, R., Swift Parrot Recovery Team, Parks and Wildlife Service, Tasmania, 1999. Swift parrot recovery plan 1997-2000. Department of Primary Industries, Water and Environment, Hobart.

Brereton, R.N., Taylor, R.J., 2000. Composition, seasonal occurrences and habitat use of bird assemblages in wet forests on the central plateau of Tasmania. Papers and Proceedings of the Royal Society of Tasmania 134, 35-43.

Brereton, R.N., Bryant, S.L., Rowell, M., 1997. Habitat modelling of the forty-spotted pardalote and recommendations for management. Report to the Tasmanian RFA Environment and Heritage Technical Committee, Hobart.

Brigham, R.M., 1991. Flexibility in foraging and roosting behaviour by the big brown bat (*Eptesicus fuscus*). Canadian Journal of Zoology 69, 117-121.

Brooker, M.I.J., Kleinig, D.A., 1999. Field guide to eucalypts. Inkata Press, Hawthorn, Victoria.

Brookhouse, M.T., 1997. Identification and analysis of growth rings in *Eucalyptus obliqua* L'Herit and *E. cypellocarpa* L. Johnson. B.Sc. (Hons.) Thesis, Australian National University, Canberra.

Brookhouse, M.T., 2006. Eucalypt dendrochronology: past, present and potential. Australian Journal of Botany 54, 435-449.

Brown, P.B., 1979. The status of parrot species in western Tasmania. Tasmanian Bird Report 9, 4-12.

Brown, P.B., 1986. The forty-spotted pardalote in Tasmania. Wildlife Division Technical Report 1986/4, Hobart.

Brown, P.B., 1989a. The swift parrot *Lathamus discolor* (White). A report on its ecology, distribution and status, including management considerations. Report for the Australian Heritage Commission and the Department of Lands Parks and Wildlife, Tasmania, Hobart.

Brown, P.B., 1989b. The forty-spotted pardalote on Maria Island. Tasmanian Bird Report 18, 4-13.

Brown, P.B., Wilson, R.I., 1982. The orange-bellied parrot. In: Groves, R.H., Ride, W.D.L. (Eds.), Species at risk: research in Australia. Proceedings of a Symposium on the Biology of Rare and Endangered Species in Australia, sponsored by the Australian Academy of Science and held in Canberra, 25 and 26 November 1981, pp. 106-115.

Brown, P.B., Wilson, R.I., 1984. Orange-bellied parrot recovery plan. National Parks and Wildlife Service, Tasmania.

Brown, P.B., Holdsworth, M.C., 1992. The status of cockatoos in Tasmania. Tasmanian Bird Report 21, 4-12.

Bryant, S., 2002. Impact of clearing old growth elements on Tasmania's woodland vertebrates. Firewood conferences, TCT, Hobart.

Bryant, S.L., 1997. Status of forty-spotted pardalote colonies. Tasmanian Bird Report 26, 45-50.

Bunnell, F.L., Wind, E., Boyland, M., Houde, I., 2002. Diameters and heights of trees with cavities: their implications to management. USDA Forest Service General Technical Report PSW-GTR-181, 717-737.

Burnham, K.P., Anderson, D.R., 2002. Model selection and multimodel inference: A practical information-theoretic approach. Springer, New York.

Calder, T.G., Golding, B.G., Manderson, A.D., 1979. Management for arboreal species in the Wombat State Forest. M. Env. Sci. Thesis, Monash University, Melbourne.

Callister, D.J., 1991. A review of the Tasmanian brushtail possum industry. TRAFFIC Bulletin 12, 49-58.

Cayley, N.W., Lendon, A.H., 1973. Australian parrots in field and aviary. Revised edition of Neville W. Cayley's original work. Angus and Robertson, Sydney.

Chambers, C.L., Mast, J.N., 2005. Ponderosa pine snag dynamics and cavity excavation following wildfire in northern Arizona. Forest Ecology and Management 216, 227-240.

Churchill, S., 1998. Australian bats. Reed New Holland, Sydney.

Commonwealth of Australia and State of Tasmania, 1997. Tasmanian Regional Forest agreement between the Commonwealth of Australia and the State of Tasmania.

Commonwealth of Australia and State of Tasmania, 2005. The Tasmanian Community Forest Agreement.

Cooper, R., 1998. A second record for Tasmania? Scepticism little lorikeets and rainbows. Tasmanian Bird Report 27, 3-4.

Coulson, R.I., Coulsen, G.M., 1981. The effect of forestry practices on bird breeding in open forest. Project Rep. 1980/4 Centre for Environmental Studies, University of Tasmania, Hobart.

Coupland, C., 2000. The distribution, roosting requirements and daily routine of white cockatoos in northern Tasmania. B.Sc. (Hons.) Thesis, University of Tasmania, Hobart.

Davey, S.M., 1984. Habitat preferences of arboreal marsupials within a coastal forest in southern New South Wales. In: Smith, A.P., Hume, I.D. (Eds.), Possums and Gliders. Surrey Beatty and Sons, Chipping Norton, NSW, pp. 509-516.

De'Ath, G., Fabricius, K.E., 2000. Classification and regression trees: A powerful yet simple technique for ecological data analysis. Ecology 81, 3178-3192.

Debus, S.J.S., 1993. The mainland masked owl *Tyto novaehollandiae*: a review. Australian Bird Watcher 15, 168-191.

Department of Natural Resources and Environment, 2002a. Central Gippsland Forest Management Area - Estimate of Sawlog resource. Department of Natural Resources and Environment, East Melbourne.

Department of Natural Resources and Environment, 2002b. Benalla-Mansfield Forest Management Area - Estimate of Sawlog resource. Department of Natural Resources and Environment, East Melbourne.

Dorr, T.L., 1999. Foraging behaviour and habitat selection of the forty-spotted pardalote, *Pardalotus quadragintus*. B.Sc. (Hons.) Thesis, University of Tasmania, Hobart.

Dove, H.S., 1910. The dusky robin (*Petroeca vittata*). Emu 10, 127-131.

Dove, H.S., 1916. Some Tasmanian birds nests. Emu 15, 234-243.

DPIW, 2001. TASVEG, The Tasmanian Vegetation Map Version 1.0. Resource Management and Conservation, Department of Primary Industries and Water, Hobart, Tasmania.

DPIW, 2006. CAR Reserves. Department of Primary Industries and Water, Hobart, Tasmania.

DPIW, 2007. Natural Values Atlas. Department of Primary Industries and Water, Hobart, Tasmania.

Drechsler, M., 1998. Spatial conservation management of the orange-bellied parrot *Neophema chrysogaster*. Biological Conservation 84, 283-292.

Driessen, M.M., Hocking, G.J., 1992. Review and analysis of spotlight surveys in Tasmania: 1975:1990. Department of Parks, Wildlife and Heritage, Tasmania.

Driessen, M.M., Taylor, R.J., Hocking, G.J., 1990. Trends in abundance of three marsupial species after fire. Australian Mammalogy 14, 121-124.

Duckworth, P., 2001. Birds in *Pinus radiata* plantation in the Fingal valley, Tasmania. Tasmanian Bird Report 29, 15.

Duhig, N., Munks, S., Wapstra, M., Taylor, R., 2000. Mortality rates of retained habitat trees in State Forest coupes: a long-term monitoring project - Initial report. Forestry Tasmania and the Forest Practices Board, Hobart.

Duncan, A.M.R., 1995. Use of silvicultural regrowth by fauna. Report to Forestry Tasmania and the Department of Primary Industries and Energy, Canberra.

Duncan, A.M.R., Taylor, R.J., 2001. Occurrence of pygmy possums, *Cercartetus lepidus* and *C. nanus*, and their nest sites in logged and unlogged dry and wet eucalypt forest in Tasmania. Australian Forestry 64, 159-164.

Duncan, F., 1999. Dry sclerophyll forests and woodlands. In: Reid, J. B., Hill, R. S., Brown, M. J. Hovenden, M. J., Vegetation of Tasmania. Environment Australia, Tasmania, pp.244-264.

Edington, J.M., Edington, M.A., 1972. Spatial patterns and habitat partitioning in the breeding birds of an upland wood. Journal of Animal Ecology 41, 331-357.

Elliott, A.J., 1931. The flame robin. Emu 30, 301-302.

Elliott, H.J., Bashford, R., 1984. Incidence and effects of the dampwood termite *Porotermes adamsoni*, in two Tasmanian east coast eucalypt forests. Australian Forestry 47, 11-15.

Fan, Z., Larsen, D.R., Shifley, S.R., Thompson, F.R., 2003. Estimating cavity tree abundance by stand age and basal area, Missouri, USA. Forest Ecology and Management 179, 231-242.

Fan, Z., Shifley, S.R., Thompson, F.R., Larsen, D.R., 2004. Simulated cavity tree dynamics under alternative timber harvest regimes. Forest Ecology and Management 193, 399-412.

Fielding, P., 1976. Birds of the far west coast of Tasmania. Tasmanian Naturalist 44, 12-16.

Fielding, P., 1979. Avifauna of Tasmanian orchards (Part 2). Tasmanian Naturalist 57, 2-12.

Fischer, J., Lindenmayer, D.B., 2002. The conservation value of paddock trees for birds in a variegated landscape in southern New South Wales. 2. Paddock trees as stepping stones. Biodiversity and Conservation 11, 833-849.

Fitzgerald, A.E., 1984. Diet of the possum (*Trichosurus vulpecula*) in three Tasmanian forest types and its relevance to the diet of possums in New Zealand Forests. In: Smith, A.P., Hume, I.D. (Eds.), Possums and Gliders. Surrey Beatty and Sons, Chipping Norton, NSW, pp. 137-143.

Fletcher, J.A., 1924. Birds of the Steppes. Emu 24, 107-117.

Forest Practices Authority, 2005. Forest Botany Manual. Forest Practices Authority, Tasmania.

Forest Practices Board, 2000. Forest Practices Code. Forest Practices Board, Hobart, Tasmania.

Forestry Commission, 1994. Silvicultural systems. Native forest silviculture Technical Bulletin No. 5, Forestry Commission, Tasmania.

Foster, 2001. Observations of wood duck (*Chenonetta jubata*) in far south and south-west Tasmania. Tasmanian Bird Report 29, 19.

Fox, J. and Brereton R., 2004. Yellow-tailed black cockatoo (*Calyptorhyncus funereus*). In J.C. Fox, T.J.Regan, S.A. Bekessy, B.A. Wintle, M.J. Brown, J.M. Meggs, K. Bonham, R. Mesibov, M.A. McCarthy, S.A. Munks, P. Wells, R. Brereton, K. Graham, J. Hickey, P. Turner, M. Jones, W.E. Brown., N. Mooney, S. Grove, K, Yamada and M.A. Burgman (Eds.). Linking landscape ecology and management to population viability analysis. Report 2: Population viability analysis for eleven forest dependant species. A project by the University of Melbourne prepared for Forestry Tasmania.

Fox, J.C., Burgman, M.A., Ades, P.K., 2001. Predictive models of hollow incidence for State Forests in central and eastern Victoria. Report for the Department of Natural Resources and Environment, Melbourne, Australia.

Frith, C.B., Frith, D.W., 1993. Notes on birds found nesting at Iron Range, Cape York peninsula, November-December 1990. Sunbird 23, 44-58.

Frith, H.J., 1967. Waterfowl in Australia. Angus and Robertson Ltd, Sydney.

Game Management Services Unit, 2005. Game tracks Issue 10. Department of Primary Industries, Water and Environment, Hobart.

Gates, J.A., 1996. The use of hollows by birds and mammals in dead river red gum and black box trees at Disher Creek, Murray River National Park, South Australia. South Australian Ornithologist 32, 65-75.

Gerhardt, R.P., 2004. Cavity nesting in raptors of Tikal National Park and Vicinity, Peten, Guatemala. Ornitologia Neotropical 15, 477-483.

Gibbons, P., 1999. Habitat tree retention in wood production forests. Ph.D. Thesis, Australian National University, Canberra.

Gibbons, P., Lindenmayer, D.B., 1996. Issues associated with the retention of hollow-bearing trees within eucalypt forests managed for wood production. Forest Ecology and Management 83, 245-279.

Gibbons, P., Lindenmayer, D.B., 1997. Developing tree retention strategies for hollow-dependent arboreal marsupials in the wood production eucalypt forests of eastern Australia. Australian Forestry 60, 29-45.

Gibbons, P., Lindenmayer, D., 2002. Tree hollows and wildlife conservation in Australia. CSIRO Publishing, Collingwood, Vic., Australia.

Gibbons, P., Lindenmayer, D.B., Barry, S.C., Tanton, M.T., 2000a. The effects of slash burning on the mortality and collapse of trees retained on logged sites in south-eastern Australia. Forest Ecology and Management 139, 51-61.

Gibbons, P., Lindenmayer, D.B., Barry, S.C., Tanton, M.T., 2000b. Hollow formation in eucalypts from temperate forests in southeastern Australia. Pacific Conservation Biology 6, 218-228.

Gibbons, P., Lindenmayer, D.B., Barry, S.C., Tanton, M.T., 2002. Hollow selection by vertebrate fauna in forests of southeastern Australia and implications for forest management. Biological Conservation 103, 1-12.

Giblin, W.W., Swindells, A.W., 1927. Field notes on the blue-winged parrot. Emu 27, 5-13.

Giese, C.L.A., Cuthbert, F.J., 2005. Woodpecker nest tree characteristics in upper midwestern oak forests. Canadian Field-Naturalist 119, 367-376.

Green, R.H., 1965. Observations on the little brown bat *Eptesicus pumilus* Gray in Tasmania. Records of the Queen Victoria Museum 20, 1-16.

Green, R.H., 1966a. Notes on Tasmanian bats. Tasmanian Naturalist Supplement 7, 1-2.

Green, R.H., 1966b. Notes on lesser long-eared bat *Nyctophilus geoffroyi* in northern Tasmania. Records of the Queen Victoria Museum 22, 1-4.

Green, R.H., 1973. The mammals of Tasmania. Published by the author, Launceston, Tasmania.

Green, R.H., 1977. The vertebrate fauna of Maggs Mountain Tasmania. Records of the Queen Victoria Museum 58, 1-40.

Green, R.H., 1979. The little pygmy possum *Cercartetus lepidus* in Tasmania. Records of the Queen Victoria Museum 68, 3-11.

Green, R.H., 1982a. The activity and movement of fauna in compartment 2, Maggs Mountain, Tasmania in the first five years of forest regeneration. Records of the Queen Victoria Museum 75, 1-31.

Green, R.H., 1982b. Breeding and food of the masked owl *Tyto novaehollandiae*. Tasmanian Naturalist 69, 4-6.

Green, R.H., 1983. The decline of the eastern rosella and other Psittaciformes in Tasmania concomitant with the establishment of the introduced European starling. Records of the Queen Victoria Museum 82, 1-5.

Green, R.H., 1995. The fauna of Tasmania. Birds. Potoroo Publishing, Launceston, Tasmania.

Green, R.H., Rainbird, J.L., 1984. The bat genus *Eptesicus Gray* in Tasmania. Tasmanian Naturalist 76, 1-5.

Green, R.H., Scarborough, T.J., McQuillan, P.B., 1988. Food and feeding of the laughing kookaburra and tawny frogmouth in Tasmania. Tasmanian Naturalist 93, 5-8.

Guiler, E.R., 1953. Distribution of the brush possum in Tasmania. Nature 172, 1091.

Gunn, G.C., 1851. On the introduction and naturalisation of *Petaurus sciureus* in Tasmania. Papers and Proceedings of the Royal Society of Tasmania 1, 253-255.

Harper, M.J., McCarthy, M.A., van der Ree, R., 2005. The abundance of hollow-bearing trees in urban dry sclerophyll forest and the effect of wind on hollow development. Biological Conservation 122, 181-192.

Harper, M.J., McCarthy, M.A., van der Ree, R., Fox, J.C., 2004. Overcoming bias in ground-based surveys of hollow-bearing trees using double-sampling. Forest Ecology and Management 190, 291-300.

Harris, J.G.K., 1980. Birds of the Sullivans Cove area, city of Hobart. Tasmanian Naturalist 60, 1-6.

Harris, S., Kitchener, A., 2005. From forest to fjaeldmark: descriptions of Tasmania's vegetation. Department of Primary Industries, Water and Environment, Printing Authority of Tasmania, Hobart.

Haseler, M., 1990. The distribution and abundance of tree hollows and their use by birds in a dry sclerophyll forest in north-east Tasmania. Report to the Forestry Commission of Tasmania, Hobart.

Haseler, M., Taylor, R., 1993. Use of tree hollows by birds in sclerophyll forest in north-eastern Tasmania. Tasforests 5, 51-56.

Hastie, T., Tibshirani, R., Friedman, J., 2003. The elements of statistical learning. Data mining, inference and prediction. Springer, New York.

Healy, W.M., Brooks, R.T., DeGraaf, R.M., 1989. Cavity trees in sawtimber-size oak stands in central Massachusetts. Northern Journal of Applied Forestry 6, 61-65.

Hickey, J.E., Wilkinson, G.R., 1999. The development and current implementation of silvicultural practices in native forests in Tasmania. Australian Forestry 62, 245-254.

Hickey, J.E., Brown, M.J., 2003. Towards ecological forestry in Tasmania. In: Lindenmayer, D.B., Franklin, J.F. (Eds.), Towards forest sustainability. CSIRO Publishing, Collingwood, Victoria, Australia, pp. 31-46.

Higgins, P.J., 1999. Handbook of Australian, New Zealand and Antarctic birds: Volume 4: parrots to dollarbird. Oxford University Press, Melbourne.

Higgins, P.J., Peter, J.M., Cowling, S.J., 2006. Handbook of Australian, New Zealand and Antarctic birds: Volume 7: boatbill to starlings, part A, boatbill to larks. Oxford University Press, Melbourne.

Hingston, A.B., 2000. Impacts of logging on autumn bird populations in the southern forests of Tasmania. Papers and Proceedings of the Royal Society of Tasmania 134, 19-28.

Hocking, G.J., 1981. The population ecology of the brush-tailed possum, *Trichosurus vulpecula* (Kerr), in Tasmania. M.Sc. Thesis, University of Tasmania, Hobart.

Hocking, G.J., 1990. The brush possum *Trichosurus vulpecula* (Kerr) in Tasmania. Report for the Department of Primary Industries, Water and Environment, Hobart.

Hocking, G.J., Guiler, E.R., 1983. The mammals of the lower Gordon river region, south-west Tasmania. Australian Wildlife Research 10, 1-23.

Hocking, G.J., Driessen, M.M., 1996. Mammals of northeast Tasmania. Records of the Queen Victoria Museum 103, 163-169.

Holdsworth, M., 2006. Breeding success and demography of the orange-bellied parrot *Neophema chrysogaster*. M.Sc. Thesis, University of Tasmania, Hobart.

Hopkins, A.J.M., Harrison, K.S., Grove, S.J., Wardlaw, T.J., Mohammed, C.L., 2006. Wood-decay fungi and saproxylic beetles associated with living *Eucalyptus obliqua* trees: early results from studies at the Warra LTER Site, Tasmania. Tasforests 16, 111-126.

Inions, G.B., 1985. The interactions between possums, habitat trees and fire. B.Sc. (Hons.) Thesis, Australian National University, Canberra.

Inions, G.B., Tanton, M.T., Davey, S.M., 1989. Effect of fire on the availability of hollows in trees used by the common brushtail possum, *Trichosurus vulpecula* Kerr, 1792, and the ringtail possum, *Pseudocheirus peregrinus* Boddaerts, 1785. Australian Wildlife Research 16, 449-458.

Jackson, J.A., Jackson, B.J.S., 2004. Ecological relationships between fungi and woodpecker cavity sites. Condor 106, 37-49.

Jackson, M.L., 1964. Soil chemical analysis. Prentice-Hall Inc. Englewood Cliffs.

Jackson, W.D., 1999. Vegetation types. In: Reid, J. B., Hill, R. S., Brown, M. J. Hovenden, M. J., Vegetation of Tasmania. Environment Australia, Tasmania, pp.1-10.

Jacobs, M.R., 1955. Growth habits of the eucalypts. Forestry and Timber Bureau, Canberra.

Johnson, K.A., 1977. Methods for the census of wallaby and possum in Tasmania, Wildlife Division Technical Report 77/2, Hobart.

Kahler, H.A., Anderson, J.T., 2006. Tree cavity resources for dependent cavity-using wildlife in West Virginia forests. Northern Journal of Applied Forestry 23, 114-121.

Kalcounis-Ruppell, M.C., Psyllakis, J.M., Brigham, R.M., 2005. Tree roost selection by bats: an empirical synthesis using meta-analysis. Wildlife Society Bulletin 33, 1123-1132.

Kavanagh, R.P., 1984. Seasonal changes in habitat use by gliders and possums in southeastern New South Wales. In: Smith, A.P., Hume, I.D. (Eds.), Possums and Gliders. Surrey Beatty and Sons, Chipping Norton, NSW, pp. 527-543.

Kavanagh, R.P., 2000. Effects of variable-intensity logging and the influence of habitat variables on the distribution of the greater glider *Petauroides volans* in montane forest, southeastern New South Wales. Pacific Conservation Biology 6, 18-30.

Kavanagh, R.P., Bamkin, K.L., 1995. Distribution of nocturnal forest birds and mammals in relation to the logging mosaic in south-eastern New South Wales, Australia. Biological Conservation 71, 41-53.

Kavanagh, R.P., Murray, M., 1996. Home range, habitat and behaviour of the masked owl *Tyto novaehollandiae* near Newcastle, New South Wales. Emu 96, 250-257.

Keisker, D.G., 2000. Types of wildlife trees and coarse woody debris required by wildlife of north-central British Columbia. Research branch of the Ministry of Forestry, Victoria, British Columbia. Working paper 50/2000.

Kirkpatrick, J., Peacock, R.J., Cullen, P.J., Neyland, M.G., 1988. The wet eucalypt forests of Tasmania. Tasmanian Conservation Trust, Hobart.

Kirkpatrick, J. B., Backhouse, S., 1989. Native trees of Tasmania. Pandani Press, West Hobart.

Lamb, D., Loyn, R.H., Smith, A.P., Wilkinson, G., 1998. Managing habitat trees in Queensland forests. A report by the Habitat Tree Technical Advisory Group to the Queensland Department of Natural Resources, Forest Resources.

Law, B.S., 1996. The ecology of bats in south-east Australian forests and potential impacts of forestry practices: a review. Pacific Conservation Biology 2, 363-374.

Legge, S., Heinshohn, R., Garnett, S., 2004. Availability of nest hollows and breeding population size of eclectus parrots, *Eclectus roratus*, on Cape York Peninsula, Australia. Wildlife Research 31, 149-161.

Lenz, M., 1990. The breeding bird communities of three Canberra suburbs. Emu 90, 145-153.

Liaw, A., Wiener, M., 2002. Classification and regression by randomForest. R News 2, 18-22.

Lindenmayer, D.B., 1994. Wildlife corridors and the mitigation of logging impacts on fauna in wood-production forests in south-eastern Australia: a review. Wildlife Research 21, 323-340.

Lindenmayer, D.B., Franklin, J.F., 2002. Conserving forest biodiversity. A comprehensive multiscaled approach. Island Press, Washington.

Lindenmayer, D.B., Norton, T.W., Tanton, M.T., 1990a. Differences between wildfire and clearfelling on the structure of montane ash forests of Victoria and their implications for fauna dependent on tree hollows. Australian Forestry 53, 61-68.

Lindenmayer, D.B., Cunningham, R.B., Donnelly, C.F., 1993a. The conservation of arboreal marsupials in the montane ash forests of the central highlands of Victoria, south-east Australia. 4. The presence and abundance of arboreal marsupials in retained linear habitats (wildlife corridors) within logged forest. Biological Conservation 66, 207-221.

Lindenmayer, D.B., Cunningham, R.B., Donnelly, C.F., 1997a. Decay and collapse of trees with hollows in eastern Australian forests: Impacts on arboreal marsupials. Ecological Applications. 7, 625-641.

Lindenmayer, D.B., Welsh, A., Donnelly, C.F., 1997b. Use of nest trees by the mountain brushtail possum (*Trichosurus caninus*) (Phalangeridae: Marsupialia). 3. Spatial Configuration and co-occupancy of nest trees. Wildlife Research 24, 661-677.

Lindenmayer, D.B., Cunningham, R.B., Tanton, M.T., Smith, A.P., 1990b. The conservation of arboreal marsupials in the montane ash forests of the central highlands of Victoria, southeast Australia. 2. The loss of trees with hollows and its implications for the conservation of Leadbeater's possum *Gymnobelideus leadbeateri* McCoy (Marsupialia, Petauridae). Biological Conservation 54, 133-145.

Lindenmayer, D.B., Welsh, A., Donnelly, C.F., Cunningham, R.B., 1996a. Use of nest trees by the mountain brushtail possum (*Trichosurus caninus*) (Phalangeridae: Marsupialia) 2. Characteristics of occupied trees. Wildlife Research 23, 531-545.

Lindenmayer, D.B., Welsh, A., Donnelly, C.F., Meggs, R.A., 1996b. Use of nest trees by the mountain brushtail possum (*Trichosurus caninus*) (Phalangeridae: Marsupialia). 1. Number of occupied trees and frequency of tree use. Wildlife Research 23, 343-361.

Lindenmayer, D.B., Cunningham, R.B., Tanton, M.T., Smith, A.P., Nix, H.A., 1990c. The conservation of arboreal marsupials in the montane ash forests of the central highlands of Victoria, southeast Australia. 1. Factors influencing the occupancy of trees with hollows. Biological Conservation. 54, 111-131.

Lindenmayer, D.B., Cunningham, R.B., Tanton, M.T., Nix, H.A., Smith, A.P., 1991. The conservation of arboreal marsupials in the montane ash forests of the central highlands of Victoria, south-east Australia. 3. The habitat requirements of

Leadbeater's possum *Gymnobelideus leadbeateri* and models of the diversity and abundance of arboreal marsupials. Biological Conservation. 56, 295-315.

Lindenmayer, D.B., Cunningham, R.B., Donnelly, C.F., Tanton, M.T., Nix, H.A., 1993b. The abundance and development of cavities in *Eucalyptus* trees - a case study in the montane forests of Victoria, southeastern Australia. Forest Ecology and Management 60, 77-104.

Lindenmayer, D.B., Cunningham, R.B., Pope, M.L., Gibbons, P., Donnelly, C.F., 2000. Cavity sizes and types in Australian eucalypts from wet and dry forest types - a simple of rule of thumb for estimating size and number of cavities. Forest Ecology and Management 137, 139-150.

Lindenmayer, D.B., Mackey, B.G., Mullen, I.C., McCarthy, M.A., Gill, A.M., Cunningham, R.B., Donnelly, C.F., 1999. Factors affecting stand structure in forests - are there climatic and topographic determinants? Forest Ecology and Management 123, 55-63.

Lloyd, P.G., Lau, J.A., 1986. Timber, vegetation and habitat resources of the Chiltern Regional Park. Department of Conservation Forests and Lands, Western Australia.

Lloyd, S., 2002. Ecological impact of the laughing kookaburra in Tasmania. Tasmanian Bird Report 30, 6-8.

Lobert, B., Lumsden, L., Brunner, H., Triggs, B., 2001. An assessment of the accuracy and reliability of hair identification of south-east Australian mammals. Wildlife Research 28, 637-641.

Lohmus, A., Remm, J., 2005. Nest quality limits the number of hole-nesting passerines in their natural cavity-rich habitat. Acta Oecologica 27, 125-128.

Lord, C., 1924. Some Tasmanian Parrots. Tasmanian Naturalist 1, 25.

Losin, N., Floyd, C.H., Schweitzer, T.E., Keller, S.J., 2006. Relationship between aspen heartwood rot and the location of cavity excavation by a primary cavity-nester, the red-naped sapsucker. Condor 108, 706-710.

Loyn, R.H., 1980. Bird populations in a mixed eucalypt forest used for production of wood in Gippsland, Victoria. Emu 80, 145-156.

Lumsden, L.F., Bennett, A.F., Silins, J.E., 2002. Location of roosts of the lesser long-eared bat *Nyctophilus geoffroyi* and Gould's wattled bat *Chalinolobus gouldii* in a

fragmented landscape in south-eastern Australia. Biological Conservation 106, 237-249.

Lumsden, L.F., Bennett, A.F., Krasna, S.P., Silins, J.E., 1995. The conservation of insectivorous bats in rural landscapes of northern Victoria. In: Bennet, A.F., Backhouse, G., Clark, T. (Eds.), People and nature conservation. Perspectives on private land use and endangered species recovery. The Royal Zoological Society of New South Wales, Mosman, Sydney, pp. 142-148.

Lunney, D., 1989. Priorities for bat conservation: Analysis of the response to a questionnaire in July 1989 by the participants of the eighth international bat research conference. Australian Zoologist 25, 71-78.

MacDonald, M.A., Apiolaza, L.A., Grove, S.J., 2005. The birds of retained vegetation corridors: A pre- and post-logging comparison in dry sclerophyll forest in Tasmania. Forest Ecology and Management 218, 277-290.

Mackowski, C.M., 1984. The ontogeny of hollows in blackbutt (*Eucalyptus pilularis*) and its relevance to the management of forests for possums, gliders and timber. In: Smith, A.P., Hume, I.D. (Eds.), Possums and Gliders. Surrey Beatty and Sons, Chipping Norton, NSW, pp. 553-567.

Mackowski, C.M., 1987. Wildlife hollows and timber management in blackbutt forest. Masters Thesis, University of New England, Armidale.

Maheswaran, B., Balasubramanian, P., 2003. Nest tree utilization by the Malabar grey hornbill *Ocyceros griseus* in the semi-evergreen forest of Mudumalai wildlife sactuary (S India). Acta Ornithologica 38, 33-37.

Manning, A.D., Lindenmayer, D.B., Barry, S.C., 2004. The conservation implications of bird reproduction in the agricultural "matrix": a case study of the vulnerable superb parrot of south-eastern Australia. Biological Conservation 120, 363-374.

Marchant, S., Higgins, P.J., 1990. Handbook of Australian, New Zealand and Antarctic birds: Volume 1: ratites to ducks: Part A ratites to petrels: Part B: Australian pelican to ducks. Royal Australasian Ornithologists Union.

Marks, G.C., Incoll, W.D., Long, I.R., 1986. Effects of crown development, branch shed and competition on wood defect in *Eucalyptus regnans* and *E. sieberi*. Australian Forest Research 16, 117-129.

Martin, K., Aitken, K.E.H., Wiebe, K.L., 2004. Nest sites and nest webs for cavity-nesting communities in interior British Columbia, Canada: Nest characteristics and niche partitioning. The Condor 106, 5-19.

Mazanec, Z., 1968. Influence of defoliation by the phasmatid *Didymuria violescens* on seasonal diameter growth and the pattern of growth rings in alpine ash. Australian Forestry 31, 3-14.

Mazurek, M.J., Zielinski, W.J., 2004. Individual legacy trees influence vertebrate wildlife diversity in commercial forests. Forest Ecology and Management 193, 321-334.

McCarthy, M.A., 2007. Bayesian methods for ecology. Cambridge University Press, Cambridge, UK.

McMahon, J.P., Hutchinson, M.F., Nix, H.A., Ord, K.D., 1995. ANUCLIM User's Guide. Centre for Resource and Environmental Studies, The Australian National University, Canberra.

Menkhorst, P.W., 1984. The application of nest boxes in research and management of possums and gliders. In: Smith, A.P., Hume, I.D. (Eds.), Possums and Gliders. Surrey Beatty and Sons, Chipping Norton, NSW, pp. 517-525.

Mollison, B.C., 1974. Birds of the Australian newsprint mills concession - Florentine and Styx valleys. Tasmanian Naturalist 37, 1-2.

Moloney, D.J., Wormington, K.R., DeStefano, S., 2002. Stag retention and use by arboreal marsupials in eucalypt forests of southeast Queensland, Australia: Implications for management. USDA Forest Service General Technical Report, Albany, C. A.

Monks, A., Efford, M.G., 2006. Selective herbivory by brushtail possums: Determining the age of ingested leaves using n-alkanes. Austral Ecology 31, 849-858.

Monterrubio-Rico, T.C., Escalante-Pliego, P., 2006. Richness, distribution and conservation status of cavity nesting birds in Mexico. Biological Conservation 128, 67-78.

Mooney, N., 1992. Diet of the masked owl in Tasmania. Tasmanian Bird Report 21, 35-55.

Mooney, N., 1997. Habitat and seasonality of nesting masked owls in Tasmania. Australian Raptor Studies 2. Birds Australia Monograph 3, 34-39.

Mooney, N.J., Johnson, K.A., 1979. Methods for the census of wallaby and possum in Tasmania. Report for National Parks and Wildlife Service, Tasmania and the Australian National Parks and Wildlife Service, Hobart.

Mucha, S.B., 1979. Estimation of tree ages from growth rings of eucalypts in northern Australia. Australian Forestry 42, 13-16.

Mueller-Dombois, D. and Ellenberg, H., 1974. Aims and methods of vegetation ecology. Wiley and Sons, New York.

Munks, S.A., 1990. Ecological energetics and reproduction in the common ringtail possum *Pseudocheirus peregrinus* (Marsupialia: Phalangeroidea). Ph.D. Thesis, University of Tasmania, Hobart.

Munks, S.A., Mooney, N., Pemberton, D., Gales, R., 2004a. An update on the distribution and status of possums and gliders in Tasmania, including off-shore islands. In: Goldingay, R.L., Jackson, S.M. (Eds.), The biology of Australian Possums and Gliders. Surrey Beatty and Sons, Chipping Norton, pp. 111-129.

Munks, S.A., Richards, K., Meggs, J.M., Brereton, R.N., 2004b. The importance of adaptive management in 'off-reserve' conservation for forest fauna: Implementing, monitoring and upgrading swift parrot *Lathamus discolor* conservation measures in Tasmania. In: Lunney, D. (Ed.) Conservation of Australia's Forest Fauna (second edition). Royal Zoological Society of New South Wales, Mosman, NSW, pp. 688-698.

Munks, S., Wapstra, M., Corkrey, R., Otley, H., Miller, G., in press. The occurrence of potential tree hollows in the dry eucalypt forests of south-eastern Tasmania, Australia. Australian Zoologist.

Napier, 1967. Birds of Mt. Elephant. Tasmanian Naturalist 10, 2-4.

Nelson, J.L., Morris, B.J., 1994. Nesting requirements of the yellow-tailed black-cockatoo, *Calyptorhynchus funereus*, in *Eucalyptus regnans* forest, and implications for forest management. Wildlife Research 21, 267-278.

Newman, M., 2002. A breeding season comparison of the Tasmanian robins. Tasmanian Bird Report 30, 19-25.

Newton, I., 1994. The role of nest sites in limiting the numbers of hole-nesting birds: a review. Biological Conservation 70, 265-276.

Norman, F.I., Brown, R.S., 1988. Aspects of the distribution and abundance of chestnut teal in south-eastern Australia. Emu 88, 70-80.

Northern NSW Forestry Services, 2000. The private hardwood forest resource in northern NSW. Produced by the Northern NSW Forestry Services for the Northern Rivers Regional Development Board. http://www.privateforestry.org.au/pri_hdw.htm.

O'Neill, M.G., 1984. Structure and composition of Tasmanian bat communities. B.Sc. (Hons.) Thesis, University of Tasmania, Hobart.

O'Neill, M.G., Taylor, R.J., 1986. Observations on the flight patterns and foraging behaviour of Tasmanian bats. Australian Wildlife Research 13, 427-432.

O'Neill, M.G., Taylor, R.J., 1989. Feeding ecology of Tasmanian bat assemblages. Australian Journal of Ecology 14, 19-31.

Orange-bellied Parrot Recovery Team, 1999. Orange-bellied parrot recovery plan 1998-2002. Department of Primary Industries, Water and Environment, Hobart, Tasmania.

Orange-bellied Parrot Recovery Team, 2006. National recovery plan for the orange-bellied parrot (*Neophema chrysogaster*). Department of Primary Industries and Water (DPIW), Hobart.

Park, P., 1981. Results from a nesting study of welcome swallows in southern Tasmania. Corella 5, 85-88.

Pattemore, V., 1980. Effects of the pulpwood industry on wildlife in Tasmania 3. Succession in bird communities and their habitats. Wildlife Division Tech. Rep. 80/1, National Parks and Wildlife Service, Hobart.

Pausas, J.G., Braithwaite, L.W., Austin, M.P., 1995. Modelling habitat quality for arboreal marsupials in the south coastal forests of New South Wales, Australia. Forest Ecology and Management 78, 39-49.

Penck, M., Torcello, J.C., Sanderson, K.J., 1995. Observations of coexistence between Adelaide and eastern rosellas (*Platycercus* spp.) in Adelaide. South Australian Ornithologist 32, 25-32.

Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., 2006. nlme: Linear and nonlinear mixed effects models 3.1-75.

Prasad, A.M., Iverson, L.R., Liaw, A., 2006. Newer classification and regression tree techniques: Bagging and random forests for ecological prediction. Ecosystems 9, 181-199.

Quinn, G.P., Keough, M.J., 2002. Experimental design and data analysis for biologists. Cambridge University Press, Cambridge.

R Development Core Team, 2006. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna (Austria).

Ratkowsky, A.V., Ratkowsky, D.A., 1977. The birds of the Mt. Wellington range, Tasmania. Emu 77, 19-22.

Rayner, M.E., 1992. Application of dendrochronology, stem analysis and inventory data in the estimation of tree and stand ages in Karri forest. Department of Conservation and Land Management, Western Australia.

Recher, H.F., 1991. The conservation and management of eucalypt forest birds: resource requirements for nesting and foraging. In: Lunney, D. (Ed.) Conservation of Australia's Forest Fauna. Royal Zoological Society of NSW, Sydney, pp. 25-34.

Recher, H.F., Thomas, D.G., Milledge, D.R., 1971. Census of dry sclerophyll. Emu 71, 40-43.

Remm, J., Lohmus, A., Remm, K., 2006. Tree cavities in riverine forests: What determines their occurrence and use by hole-nesting passerines? Forest Ecology and Management 221, 267-277.

Rhodes, M.P., 1996. Use of silvicultural regrowth for foraging by Tasmanian bat species. Report to Forestry Tasmania and the Department of Primary Industry and Energy, Canberra, Hobart, Tasmania.

Robinson, D., 1990. The nesting ecology of sympatric scarlet robin *Petroica multicolor* and flame robin *P. phoenicea* populations in open eucalypt forest. Emu 90, 40-52.

Rose, P.W., 1993. Production of habitat hollows by wheatbelt eucalypts. Report for the Department of Conservation and Land Management, Western Australia.

Rounsevell, D.E., Taylor, R.J., Hocking, G.J., 1991. Distribution records of native terrestrial mammals in Tasmania. Wildlife Research 18, 699-717.

Rowley, I., 1990. Bahavioural ecology of the galah *Eolophus roseicapillus* in the wheatbelt of Western Australia, Surrey Beatty and Sons Pty. Ltd, Chipping Norton, NSW.

Ruczynski, I., Bogdanowicz, W., 2005. Roost cavity selection by *Nyctalus noctula* and *N. leisleri* (Vespertilionidae, Chiroptera) in Bialowieza primeval forest, eastern Poland. Journal of Mammalogy 86, 921-930.

Saunders, D.A., Smith, G.T., Rowley, I., 1982. The availability and dimensions of tree hollows that provide nest sites for cockatoos (Psittaciformes) in Western Australia. Australian Wildlife Research 9, 541-556.

Savva, N., Taylor, R., 1986. Bat remains in a Tasmania cave. Macroderma 2, 21.

Sharland, M., 1947. Notes on two *Neophema* parrots. Emu 46, 258-264.

Sharland, M., 1958. Tasmanian birds. Angus and Robertson, Sydney.

Sharland, M., 1983. Pardalotes in and near Hobart. Tasmanian Naturalist 72, 2-3.

Slater, J., 1987a. Arboreal marsupial fauna of southern Tasmania. 1. The species and their distribution, The author, Sandy Bay, Tasmania.

Slater, J.E., 1987b. The autumn/winter diet of the sugar glider, *Petaurus breviceps*, with an investigation of Banksia nectar as a food source. University of Tasmania, Hobart.

Slater, P., 1962. Breeding birds in the lower wongong valley. Western Australian Naturalist 8, 124-126.

Smith, A.P., 1993. Habitat tree retention in the Wingham Management Area. Report to the Department of Planning, Department of Ecosystem Management, University of New England, Armidale.

Smith, A.P., Lindenmayer, D.B., 1988. Tree hollow requirements of Leadbeater's possum and other possums and gliders in timber production Ash forests of the Victorian Central Highlands. Australian Wildlife Research 15, 347-362.

Smith, A.P., Lindenmayer, D.B., 1992. Forest succession and habitat management for Leadbeater's possum in the state of Victoria, Australia. Forest Ecology and Management 49, 311-332.

Smith, A.P., Murray, M., 2003. Habitat requirements of the squirrel glider (*Petaurus norfolcensis*) and associated possums and gliders on the New South Wales central coast. Wildlife Research 30, 291-301.

Smith, A.P., Lindenmayer, D.B., Suckling, G.C., 1985. The ecology and management of Leadbeater's Possum. Research Report to the World Wildlife Fund, University of New England, Armidale, NSW.

Smith, D.T., 1998. Mapping the bushfire danger of Hobart. Grad. Dip. Thesis, University of Tasmania, Hobart.

Smith, G., Lees, N., 1998. Density and distribution of habitat trees required to support viable populations of hollow-dependent species. Department of Natural Resources, Brisbane.

Smith, P., 1985. Effects of intensive logging on birds in eucalypt forest near Bega, New South Wales. Emu 85, 15-21.

Soderquist, T.R., Lee, A.K., 1994. The ontogeny of hollows in Box-Ironbark forest. Department of Conservation and Natural Resources. Australis Pty. Ltd.

Spiegelhalter, D., Thomas, A., Best, N., Lunn, D., 2003. WinBUGS User Manual. MRC Biostatistics Unit, Cambridge, UK.

Spiegelhalter, D.J., Best, N.G., Carlin, B.P., Van der Linde, A., 2002. Bayesian measures of model complexity and fit. Journal of the Royal Statistical Society B 64, 583-639.

SPSS, 2005. SPSS for Windows 14.0, Rel. 12.0.1. SPSS Inc., Chicago.

Statham, H., Statham, H.L., 1997. Movements and habits of brushtail possums (*Trichosurus vulpecula* Kerr) in an urban area. Wildlife Research 24, 715-726.

Statham, H.L., 1984. The effects of forest operations on wildlife in Tasmania. A literature review submitted to the Forest Ecology Research Fund, Hobart.

Statham, H.L., 1987. The fauna of the southern forests. Forestry commission, Tasmania.

Stone, M.G., 1998. Forest-type mapping by photo-interpretation: A multi-purpose base for Tasmania's forest management. Tasforests 10, 15-32.

Stoneman, G.L., Rayner, M.E., Bradshaw, F.J., 1997. Size and age parameters of nest trees used by four species of parrot and one species of cockatoo in south-west Australia: Critique. Emu 97, 94-96.

Swift Parrot Recovery Team, 2001. Swift parrot recovery plan. Department of Primary Industries, Water and Environment, Hobart.

Tarburton, M.K., 1993. A comparison of the breeding biology of the welcome swallow in Australia and recently colonised New Zealand. Emu 93, 34-43.

Tasmanian Public Land Use Commission, 1996. Tasmanian-Commonwealth Regional Forest Agreement Environment and Heritage Report Vol II Background Report part C. , Tasmanian Public Land Use Commission in conjunction with Commonwealth Forests Taskforce, Hobart.

Taylor, R.J., 1991. Fauna conservation in production forests in Tasmania. Forestry Commission, Hobart.

Taylor, R.J., O'Neill, M.G., 1985. Composition of the bat (Chiroptera: Vespertilionidae) communities in Tasmanian forests. Australian Mammalogy 9, 125-130.

Taylor, R.J., Savva, N.M., 1988. Use of roost sites by four species of bats in State Forest in south-eastern Tasmania. Australian Wildlife Research 15, 637-645.

Taylor, R.J., Haseler, M., 1993. Occurrence of potential nest trees and their use by birds in sclerophyll forest in north-east Tasmania. Australian Forestry 56, 165-171.

Taylor, R.J., Comfort, M.D., 1993. Small terrestrial mammals and bats of Melaleuca and Claytons, southwestern Tasmania. Papers and Proceedings of the Royal Society of Tasmania 127, 33-37.

Taylor, R.J., Haseler, M.E., 1995. Effects of partial logging systems on bird assemblages in Tasmania. Forest Ecology and Management 72, 131-149.

Taylor, R.J., O'Neill, M.G., Reardon, T., 1987. Tasmanian bats: identification, distribution and natural history. Papers and Proceedings of the Royal Society of Tasmania 121, 109-119.

Taylor, R.J., Bryant, S.L., Pemberton, D., Norton, T.W., 1985. Mammals of the upper henty river region, western Tasmania. Papers and Proceedings of the Royal Society of Tasmania 119, 7-14.

Taylor, R.J., Duckworth, P., Johns, T., Warren, B., 1997. Succession in bird assemblages over a seven-year period in regrowth dry sclerophyll forest in south-east Tasmania. Emu 97, 220-230.

Therneau, T.M., Atkinson, B., Ripley, B., Oksanen, J., De'Ath, G., 2006. mvpart: Multivariate partitioning 1.2-4.

Thomas, D., 1979. Tasmanian Bird Atlas. Fauna of Tasmania Handbook No. 2. University of Tasmania, Hobart.

Thomas, D.G., 1972. Birds of a suburban garden. Tasmanian Naturalist 31, 5-8.

Thomas, H.F., 1957. The starling in the Sunraysia district, Victoria. Emu 57, 325-337.

Threatened Species Section, 2006. Fauna recovery plan: Forty-spotted pardalote 2006-2010, Department of Primary Industries and Water, Hobart.

Tidemann, C.R., Flavel, S.C., 1987. Factors affecting choice of diurnal roost site by tree-hole bats (Microchiroptera) in south-eastern Australia. Australian Wildlife Research 14, 459-473.

Walker, J.S., Cahill, A.J., Marsden, S.J., 2005. Factors influencing nest-site occupancy and low reproductive output in the critically endangered yellow-crested cockatoo *Cacatua sulphurea* on Sumba, Indonesia. Bird Conservation International 15, 347-359.

Wall, L.E., 1985. Little pygmy possum *Cercartetus lepidus*. Tasmanian Naturalist 8-9, 10-11.

Walsh, C., MacNally, R., 2005. hier.part: Hierarchical partitioning 1.0-1.

Walshe, T.V., 2001. Describing natural disturbance regimes for use as a forest management tool. Ph.D. Thesis, University of Melbourne, Melbourne.

Walter, S.T., Maguire, C.C., 2005. Snags, cavity-nesting birds, and silvicultural treatments in Western Oregon. Journal of Wildlife Management 69, 1578-1591.

Wapstra, M., Taylor, R.J., 1998. Use of retained trees for nesting by birds in logged eucalypt forest in north-eastern Tasmania. Australian Forestry 61, 48-52.

Wapstra, M., Doran, N., 2004. Observations on a nesting hollow of yellow-tailed black cockatoo, and the felled tree that hosted it, in north-eastern Tasmania. Tasmanian Naturalist 126, 59-63.

Wapstra, M., Thompson, V., Hudler, P., 2000. Observations of echidnas using tree hollows in Tasmanian forests. Tasmanian Naturalist 122, 29-31.

Ward, S.J., 1990. Life history of the eastern pygmy possum, *Cercatetus nanus* (Acrobatidae, Marsupialia) in south-eastern Australia. Australian Journal of Zoology 38, 287-304.

Waters, J.R., Noon, B.R., Verner, J., 1990. Lack of nest site limitation in a cavity-nesting bird community. Journal of Wildlife Management 54, 239-245.

Watts, D., 1993. Tasmanian Mammals- A field guide. Peregrine Press, Tasmania.

Wayne, A., Kavanagh, R.P., Loyn, R.H., Munks, S.A., Smith, G., 2006. Brief summary of prescriptions for the retention of hollow bearing trees in multiple-use forests throughout Australia. Report for the Research Priorities and Coordination Committee.

Wayne, A.F., 2005. The ecology of the Koomal (*Trichosurus vulpecula hypoleucus*) and Ngwayir (*Pseudocheirus occidentalis*) in the Jarrah forests of south-western Australia. Ph.D. Thesis, Australian National University, Canberra.

Webb, J.K., Shine, R., 1997. Out on a limb: conservation implications of tree-hollow use by a threatened snake species (*Hoplocephalus bungaroides*: Serpentes, Elapidae). Biological Conservation 81, 21-33.

Webb, M., Holdsworth, M., Voogdt, J., 2007. Nesting requirements of the swift parrot *Lathamus discolour*. Unpublished report to the Department of Primary Industries and Water, Hobart.

Wells, P., Hickey, J., 1999. Wet sclerophyll, mixed and swamp forest. In: Reid, J. B., Hill, R. S., Brown, M. J. Hovenden, M. J., Vegetation of Tasmania. Environment Australia, Tasmania, pp.224-243.

Welsh, A.H., Lindenmayer, D.B., Donnelly, C.F., Ruckstuhl, A., 1998. Use of nest trees by the mountain brushtail possum (*Trichosurus caninus*) (Phalangeridae : Marsupialia). IV. Transitions between den trees. Wildlife Research 25, 611-625.

Whiteley, S.B., 1999. Calculating the sustainable yield of Tasmania's State Forests. Tasforests 11, 23-34.

Whitford, K.R., 2002. Hollows in jarrah (*Eucalyptus marginata*) and marri (*Corymbia calophylla*) trees I. Hollow sizes, tree attributes and ages. Forest Ecology and Management 160, 201-214.

Whitford, K.R., Williams, M.R., 2001. Survival of jarrah (*Eucalyptus marginata* Sm.) and marri (*Corymbia calophylla* Lindl.). Habitat trees retained after logging. Forest Ecology and Management 146, 181-197.

Whitford, K.R., Williams, M.R., 2002. Hollows in jarrah (*Eucalyptus marginata*) and marri (*Corymbia calophylla*) trees II. Selecting trees to retain for hollow dependent fauna. Forest Ecology and Management 160, 215-232.

Wilson, R.I., 1981. The woodchip industry and Tasmanian birds. Tasmanian Bird Report 11, 11-15.

Wilson, R.I., 1984. Breeding habitat requirements of four endemic bird species in Tasmanian dry sclerophyll forests: green rosella, dusky robin, yellow wattlebird and swift parrot. Unpublished report to the Tasmanian National Parks and Wildlife Service, Hobart.

Woinarski, J.C.Z., Rounsevell, D.E., 1983. Comparative ecology of pardalotes, including the forty-spotted pardalote, *Pardalotus quadragintus* (Aves: Pardalotidae) in south-eastern Tasmania. Australian Wildlife Research 10, 351-361.

Woinarski, J.C.Z., Bulman, C., 1985. Ecology and breeding biology of the forty-spotted pardalote and other pardalotes on north Bruny Island. Emu 85, 106-120.

Wood, S., 2006. mgcv: GAMs with GCV smoothness estimation and GAMMs by REML / PQL 1.3-19.

Woodgate, P.W., Peel, W.D., Ritman, K.T., Coram, J.E., Brady, A., Rule, A.J., Banks, J.C.G., 1994. A study of the old-growth forests of East Gippsland, Department of Conservation and Natural Resources, Melbourne.

Wormington, K., Lamb, D., 1999. Tree hollow development in wet and dry sclerophyll eucalypt forest in south-east Queensland, Australia. Australian Forestry 62, 336-345.

Wormington, K.R., Lamb, D., McCallum, H.I., Moloney, D.J., 2002. Habitat requirements for the conservation of arboreal marsupials in dry sclerophyll forests of southeast Queensland, Australia. Forest Science 48, 217-227.

Wormington, K.R., Lamb, D., McCallum, H.I., Moloney, D.J., 2003. The characteristics of six species of living hollow-bearing trees and their importance for arboreal marsupials in the dry sclerophyll forests of southeast Queensland, Australia. Forest Ecology and Management 182, 75-92.

Appendix 1. Results of a survey to gather information on the use of tree hollows by birds in Tasmania

Introduction

Australia has a large number of species which use tree hollows for nesting or roosting (Gibbons and Lindenmayer, 2002), but has no primary excavators such as woodpeckers. This means that all hollows are produced by slow processes, generally involving fire, fungi and termites, (although some species are known to modify the size of cavities to an extent, e.g. cockatoos, brushtail possums: Ambrose, 1982; Saunders *et al.*, 1982). Consequently, many years are required to form hollows, especially large hollows. There is a general decline in hollow-bearing trees across Australia due to land clearing for urbanization and agriculture, forestry activities and the dying out of hollow-bearing trees retained in paddocks and urban areas (Gibbons and Lindenmayer, 2002). This has resulted in concern for hollow-using fauna across the country (Lindenmayer *et al.*, 1993b; Gibbons and Lindenmayer, 1997; Whitford and Williams, 2002; Wormington *et al.*, 2002).

In Tasmania, hollow dependent fauna are a management priority under the Tasmanian Regional Forest Agreement (Commonwealth of Australia and State of Tasmania, 1997). Forest management agencies in most states of Australia, including Tasmania, have developed management prescriptions for the conservation of habitat for hollow-using fauna (Wayne *et al.*, 2006). Yet the hollow requirements and the degree to which fauna are dependent on hollows vary greatly between species. Consequently, an essential element of any retention strategy is knowledge of the fauna that use hollows in the region and their known or likely hollow requirements (Recher, 1991). There are large differences in the amount of literature available between Tasmanian fauna species, birds in particular, with more information generally available for threatened species. This paper presents the results of a survey distributed to members of Birds Tasmania, intended to gather anecdotal information to assist in assessing the degree to which Tasmania's bird fauna are dependent on tree hollows. The information collected can also be used to help assess the conservation status and threatening process for these species.

Methods

A survey was distributed to all receivers (approximately 320) of the Birds Tasmania newsletter. The survey asked four questions about 38 bird species found in Tasmania. The questions asked and the potential responses are outlined in Table A1.1. The questions aimed to gather the opinions of the survey participants on the degree to which the species used tree hollows, the population status of the species and the processes threatening populations of the species. The species considered in the questionnaire were those where mention was found in the literature that they use tree hollows (e.g. Sharland, 1958; Munks *et al.*, in press). Any additional comments on the status, population size and distribution, use of tree hollows or preferred habitat by the species included in the survey were also invited.

Table A1.1 Questions asked for each bird species included in the survey.

Question	Potential Responses
How do you rate your knowledge of this species and its current status?	Expert High Average Low None
How do you rate the hollow dependency of this species?	Reliant on hollows for roosting and breeding Reliant on hollows for breeding but not roosting Use hollows for roosting and/or breeding but can use other sites Do not use hollows
How do you rate the status of populations of this species?	Increasing Stable at high numbers Stable at low numbers Declining
Of the following issues, circle any you believe to be of concern to this species	Forestry activities Land clearing for agriculture Competition for nest sites with bees or introduced birds Over predation by endemic or introduced species Road-kill Hunting There are no concerns

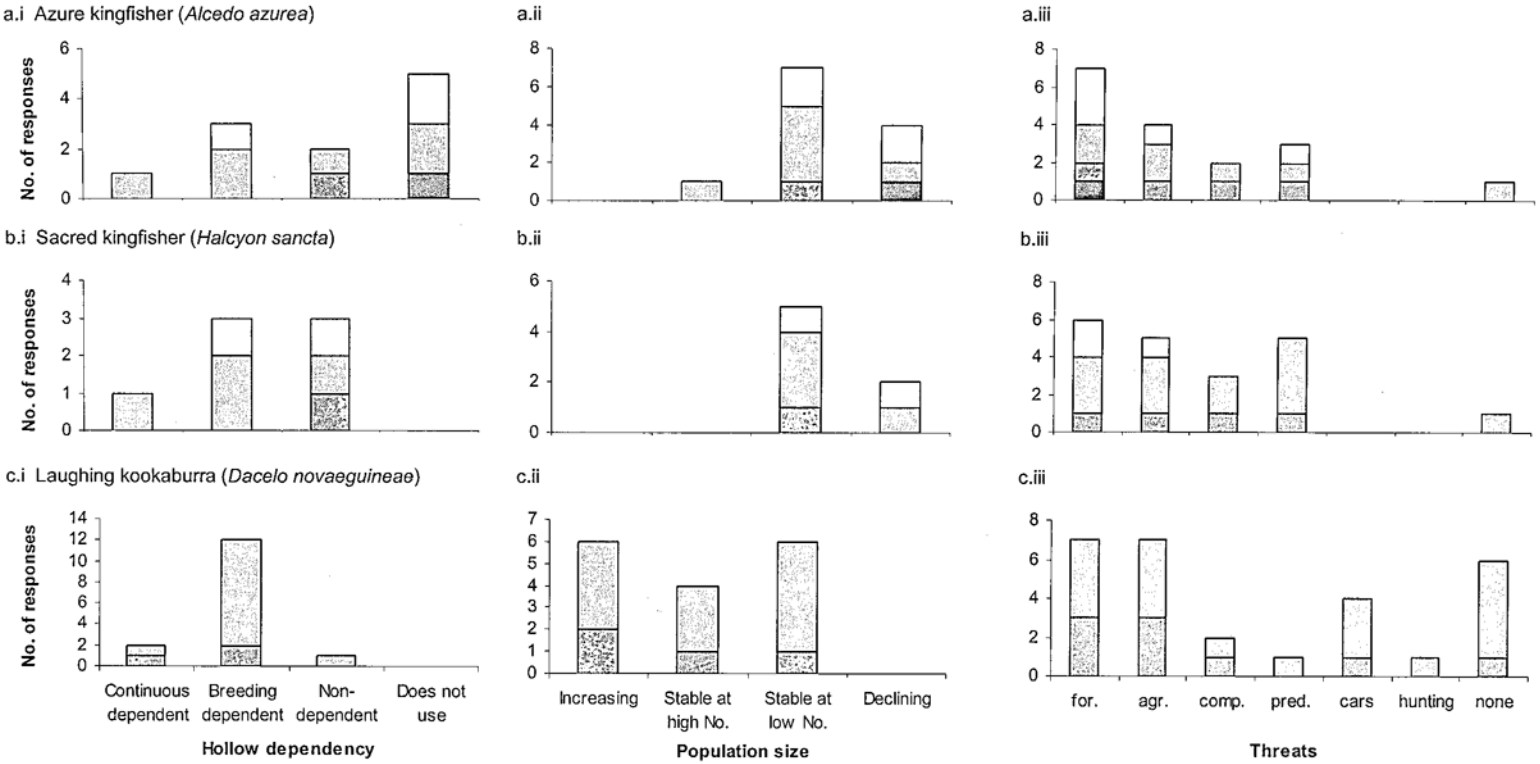


Figure A1.1. (a – c) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

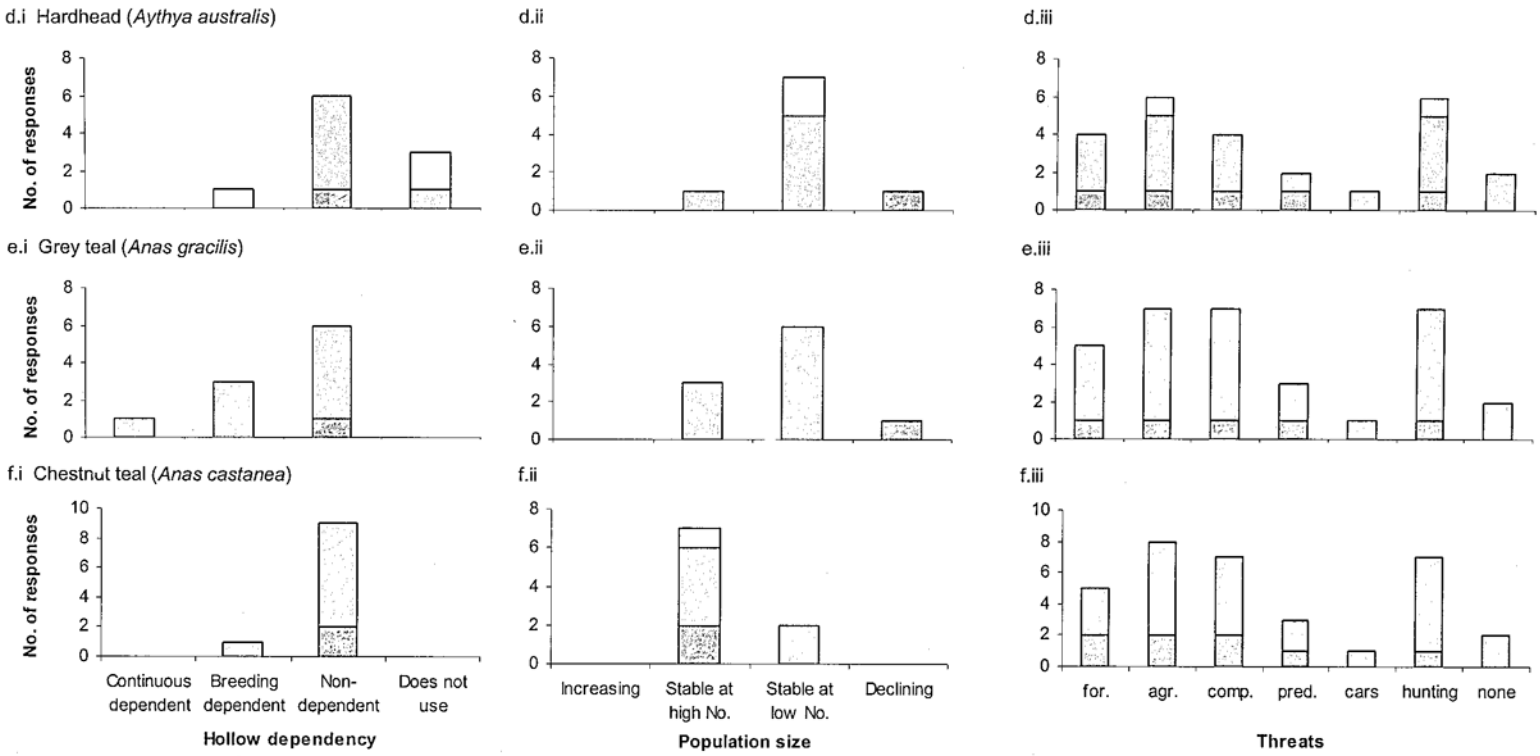


Figure A1.1. (d – f) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

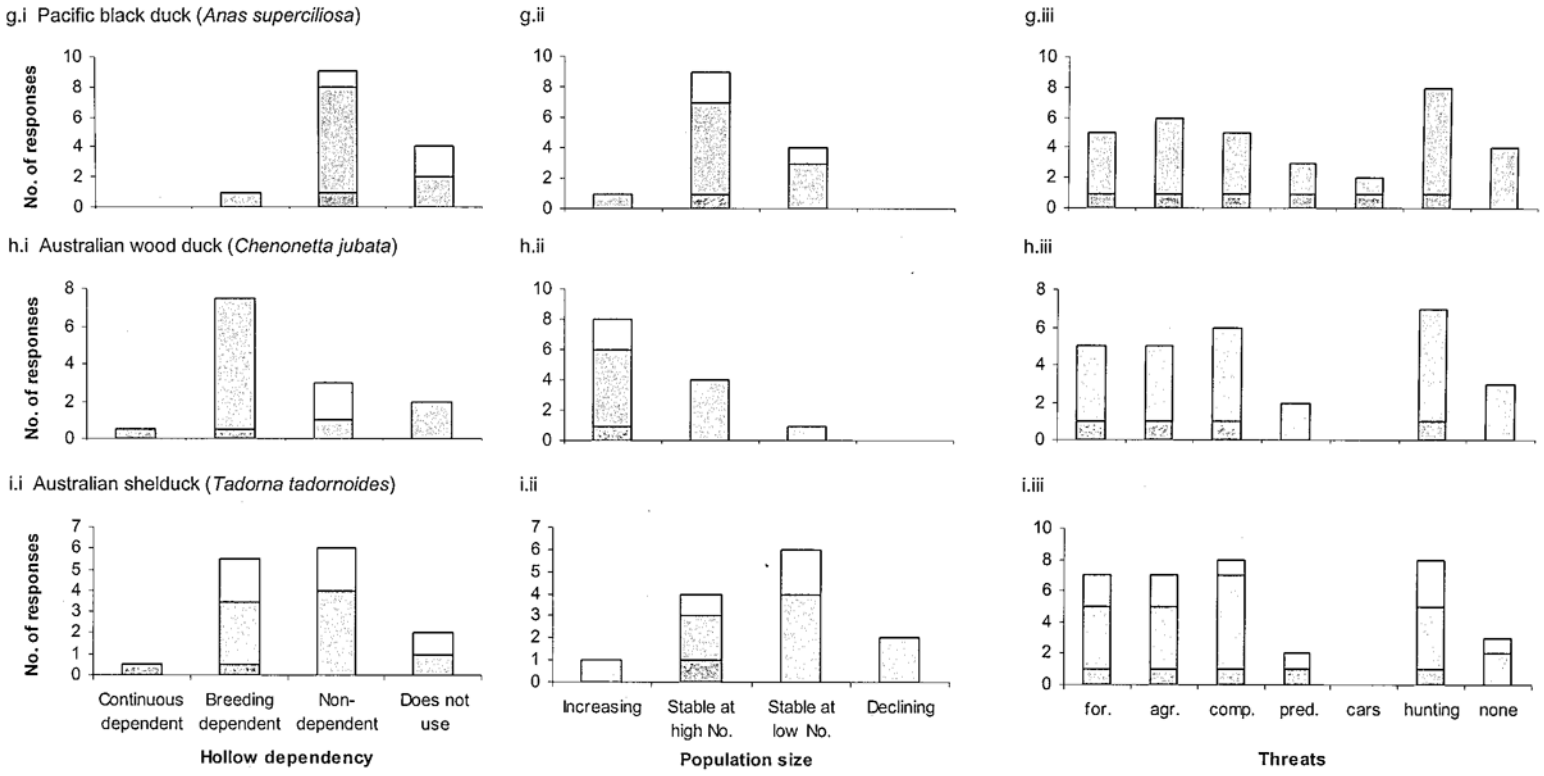


Figure A1.1. (g – i) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

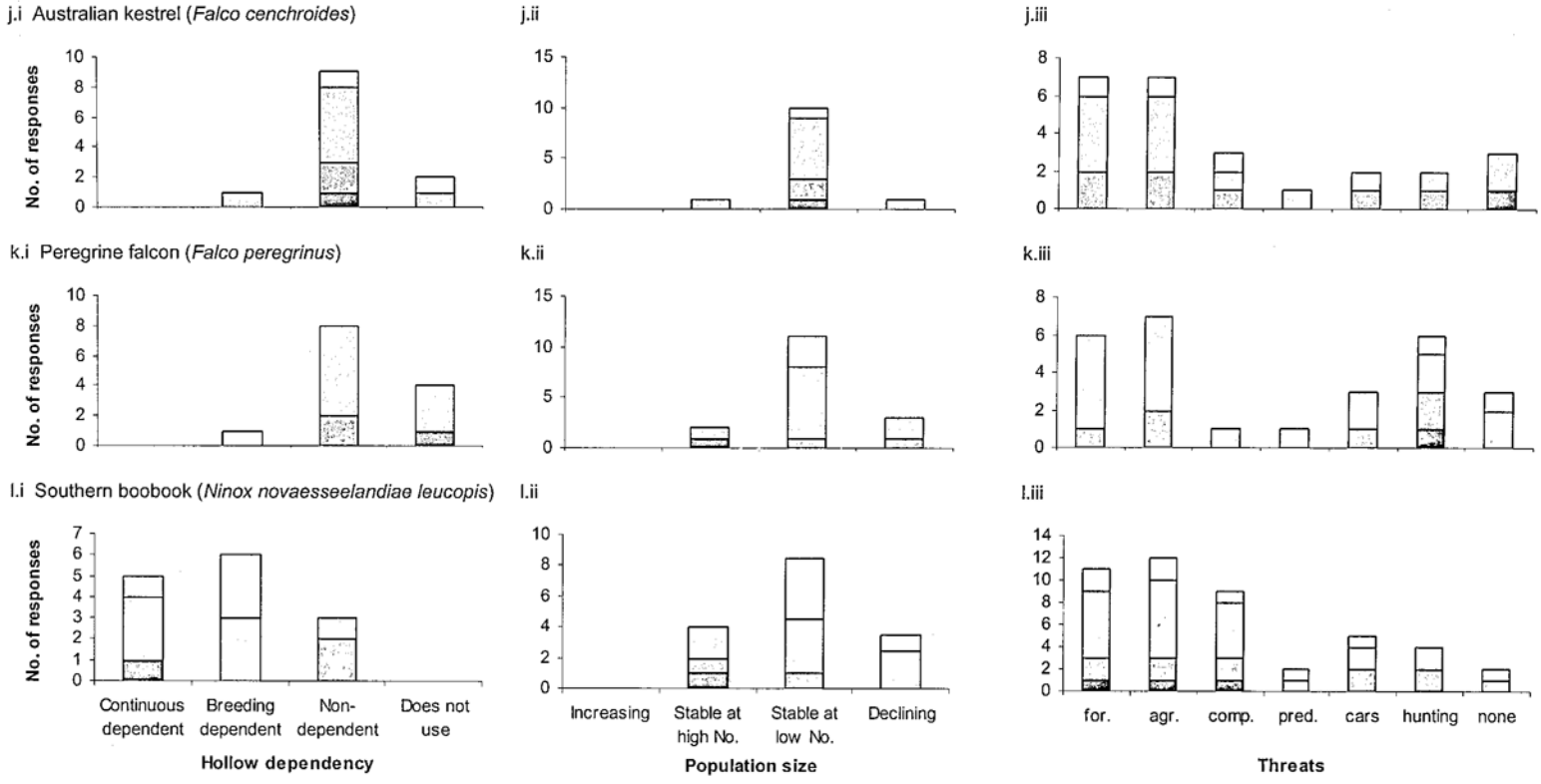


Figure A1.1. (j - l) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

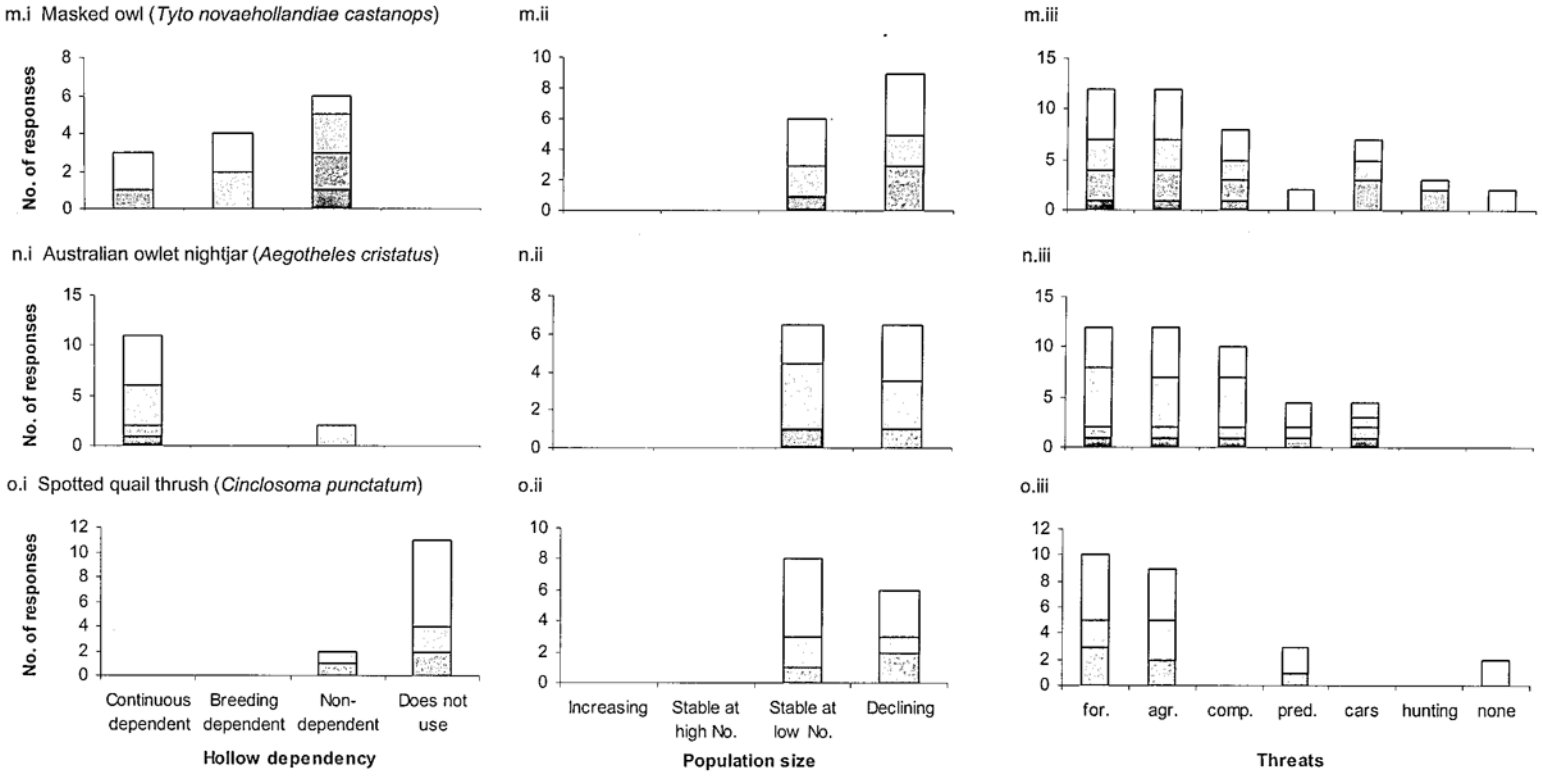


Figure A1.1. (m – o) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

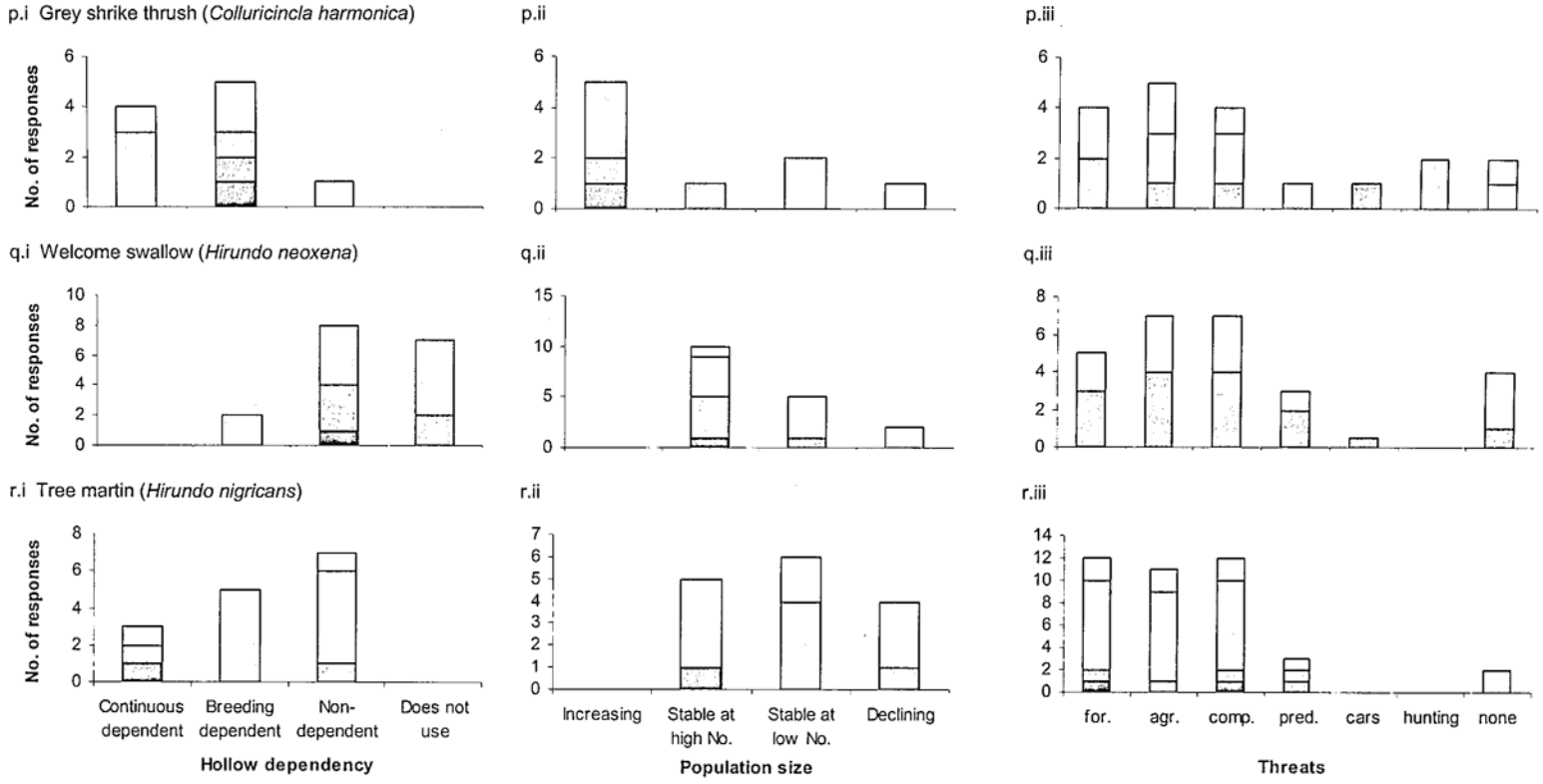


Figure A1.1. (p – r) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

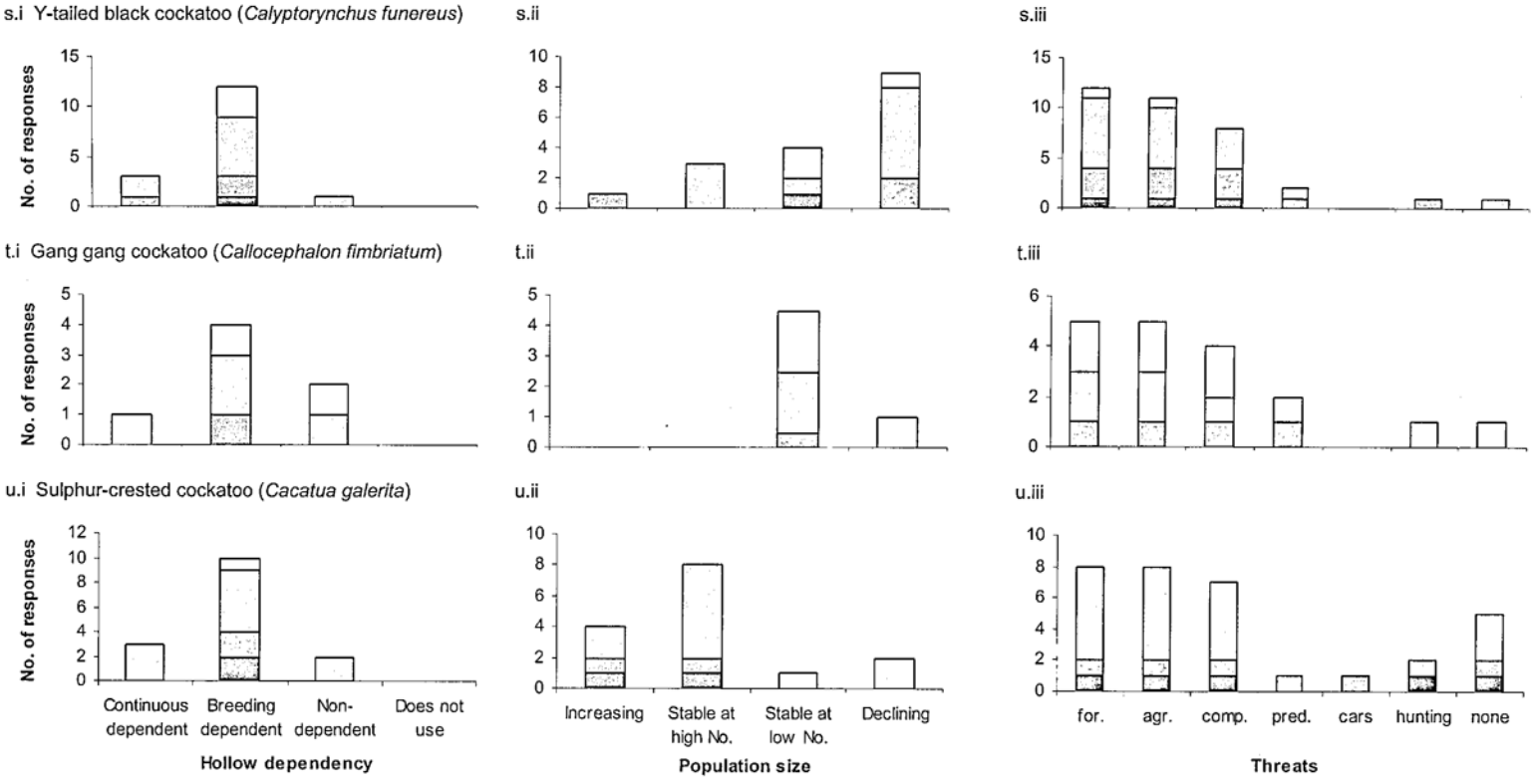


Figure A1.1. (s – u) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

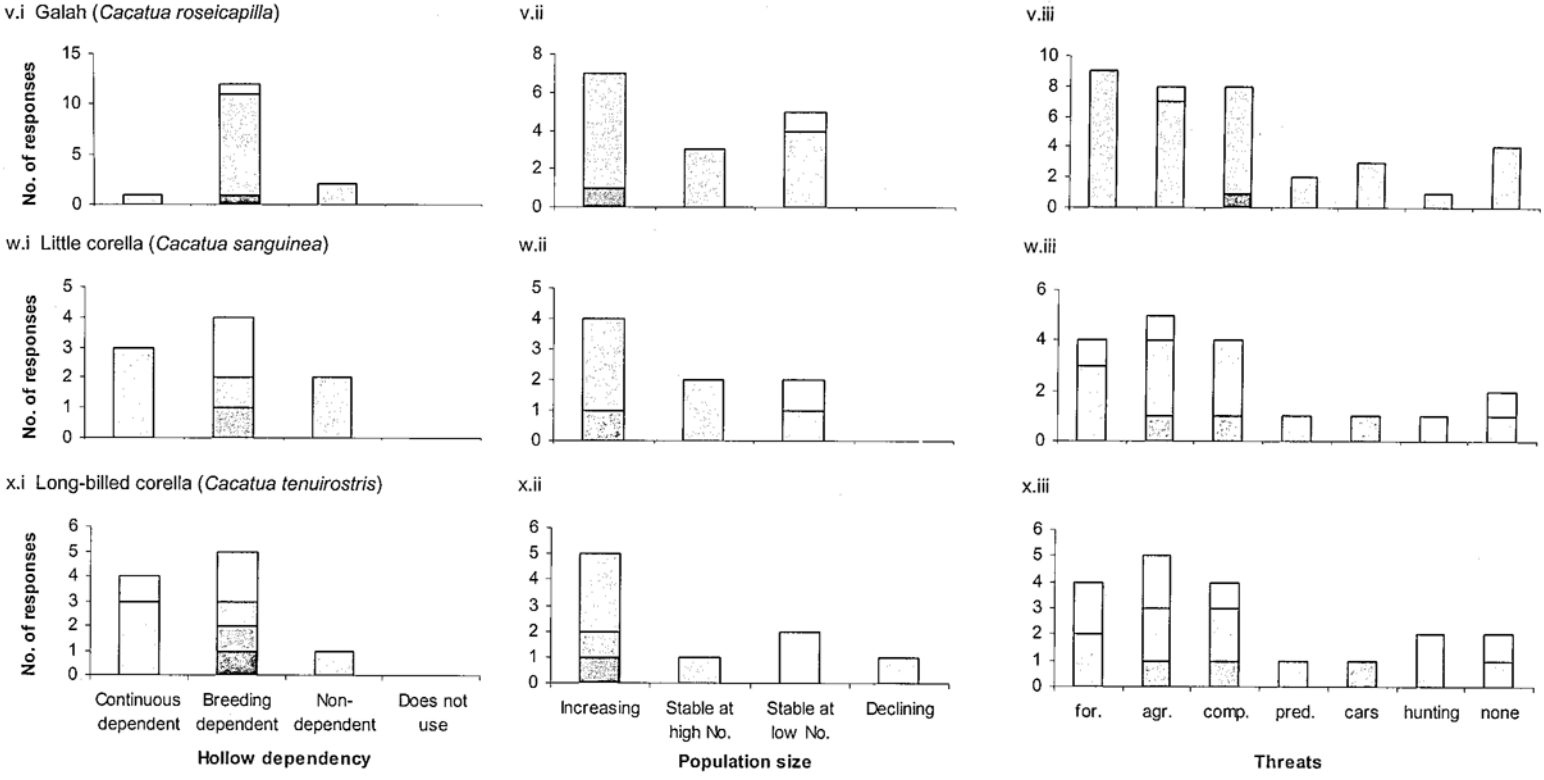


Figure A1.1. (v – x) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

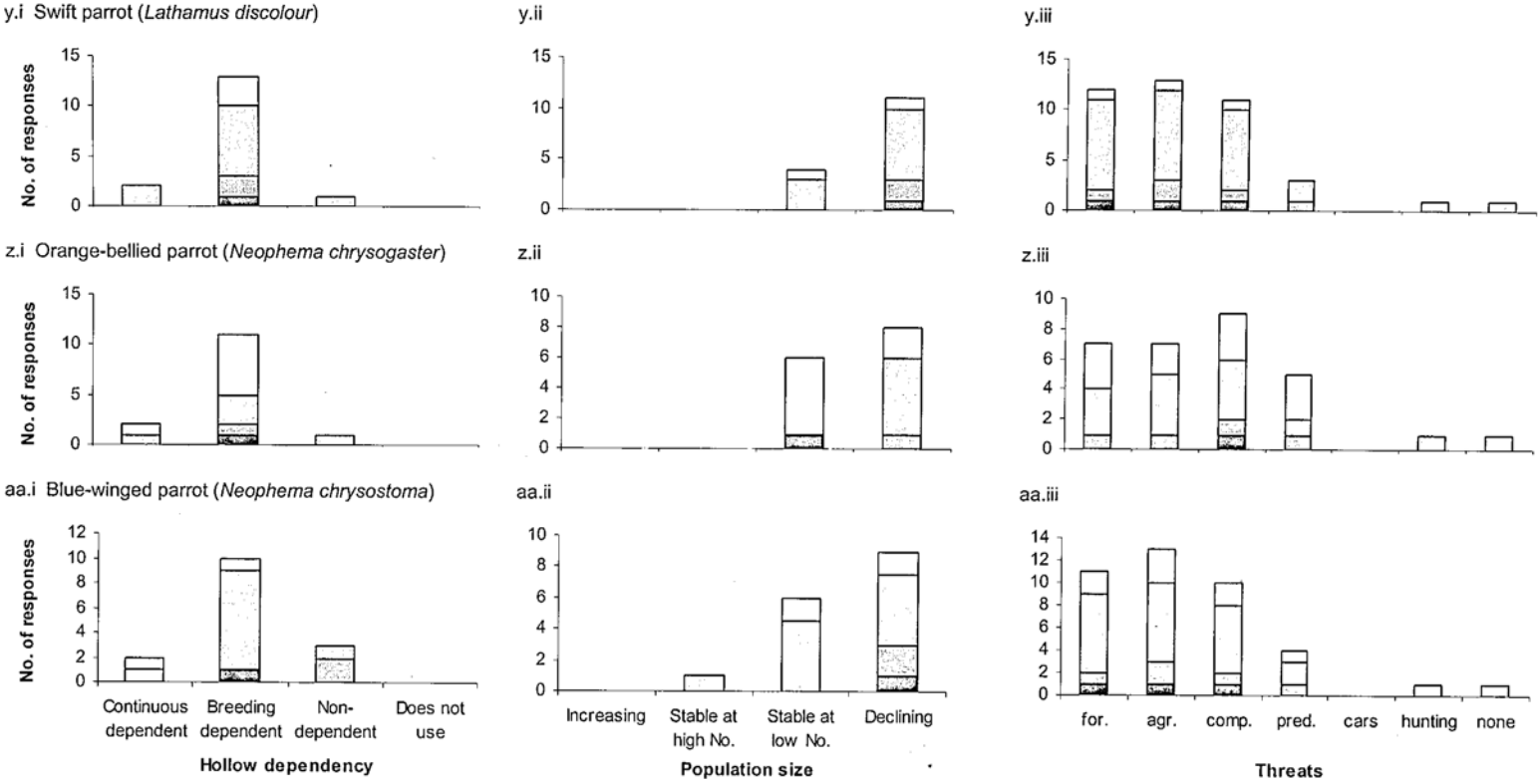


Figure A1.1. (y – aa) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

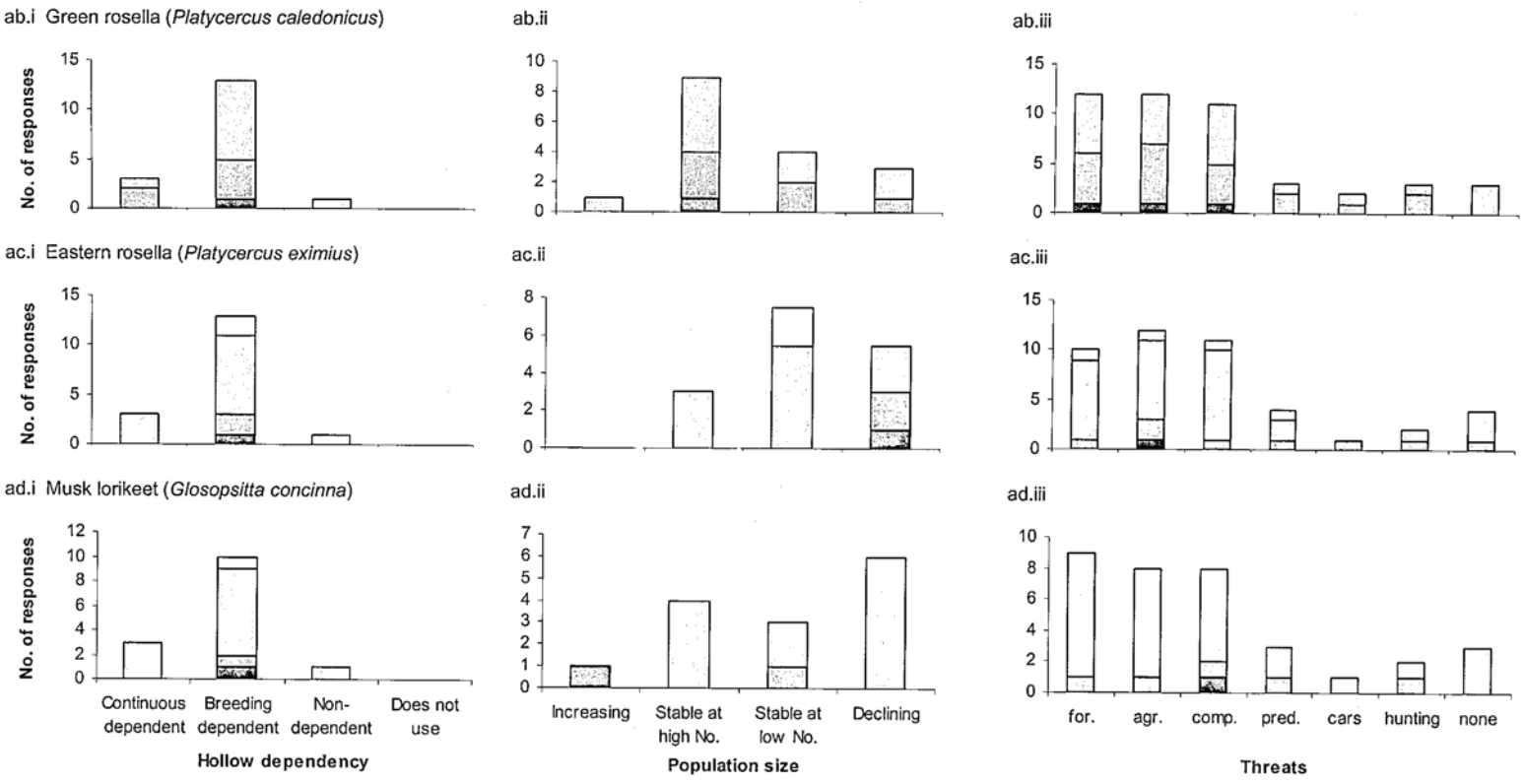


Figure A1.1. (ab – ad) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

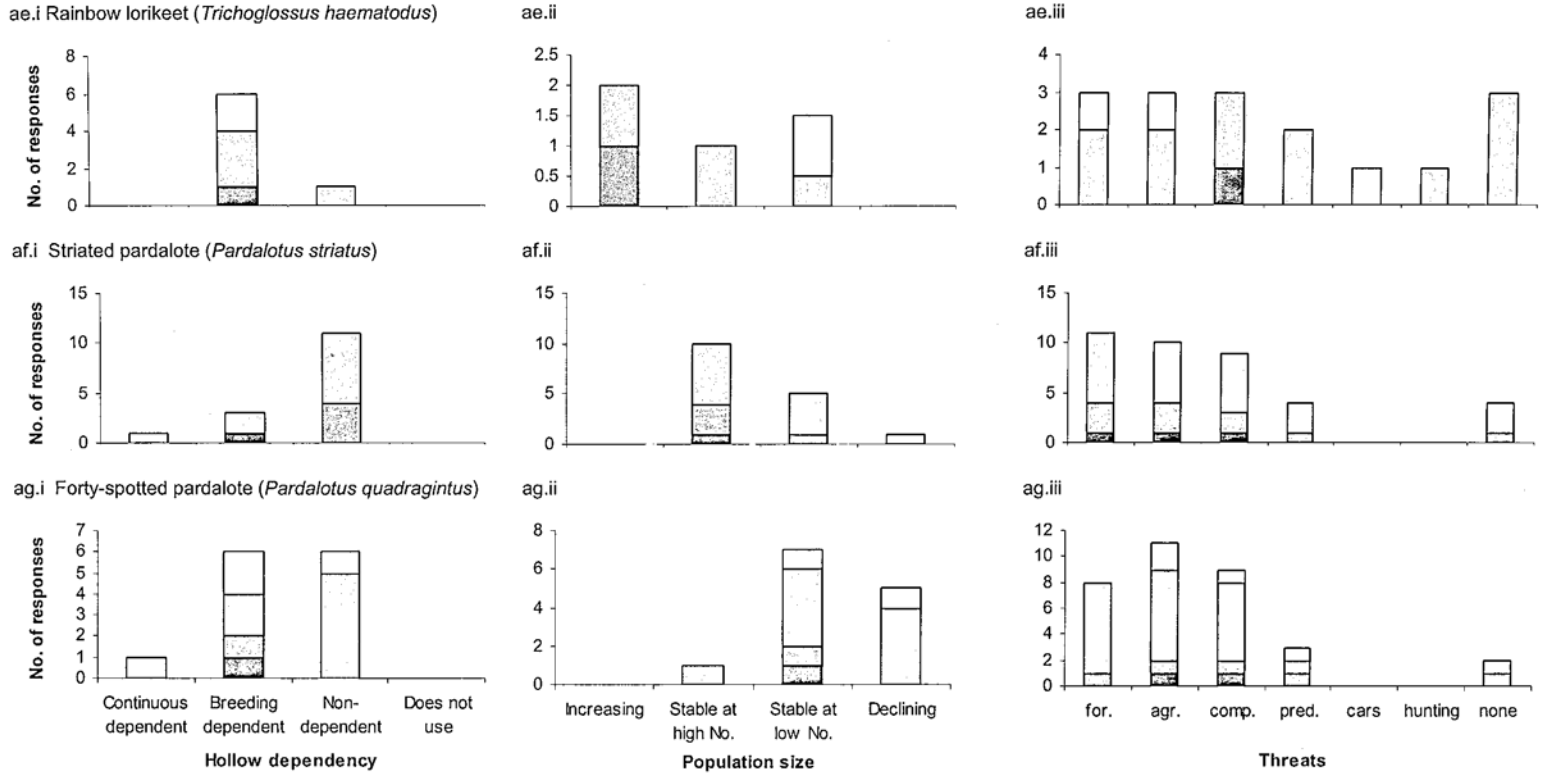


Figure A1.1. (ae – ag) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

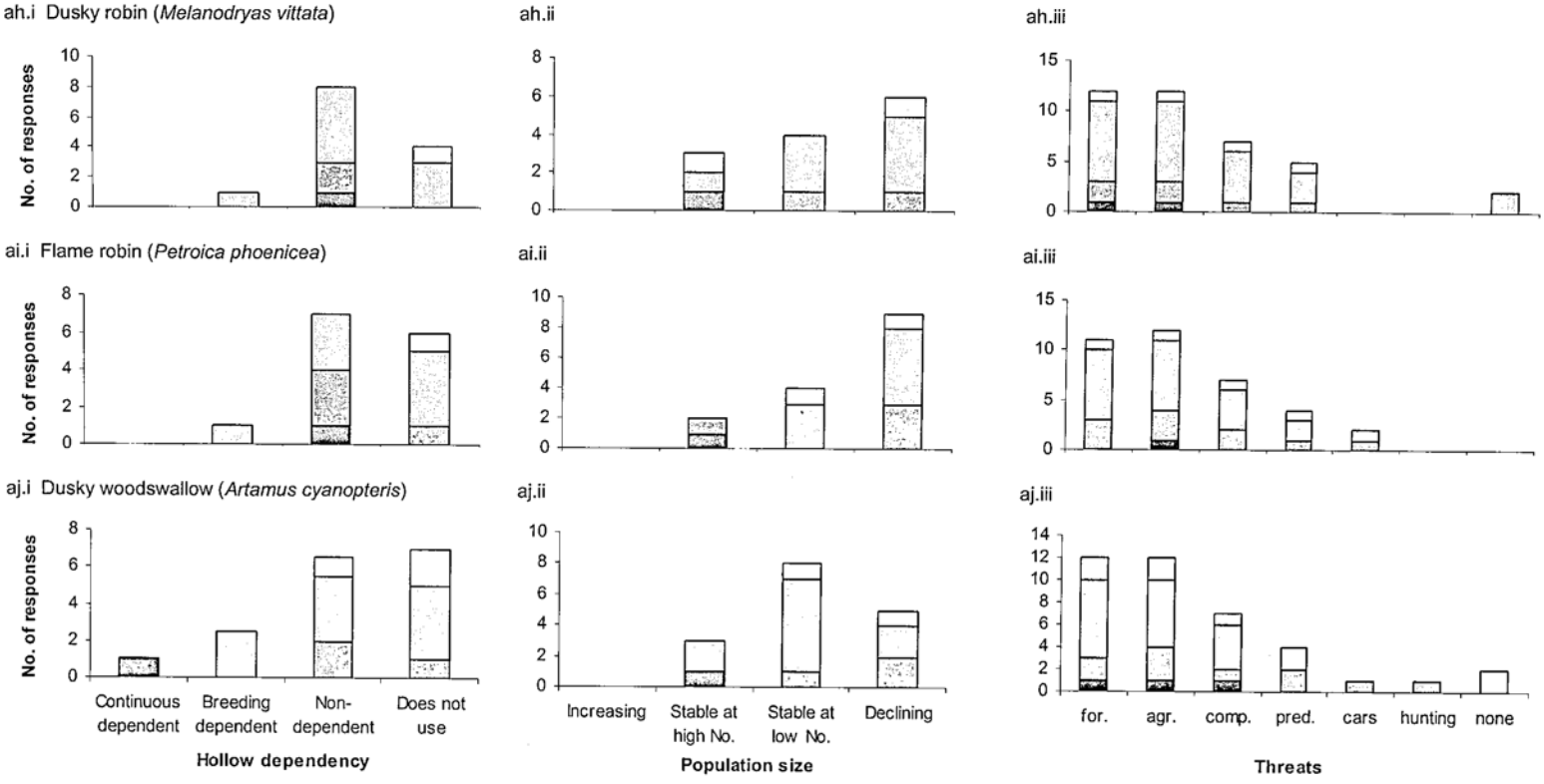


Figure A1.1. (ah – aj) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

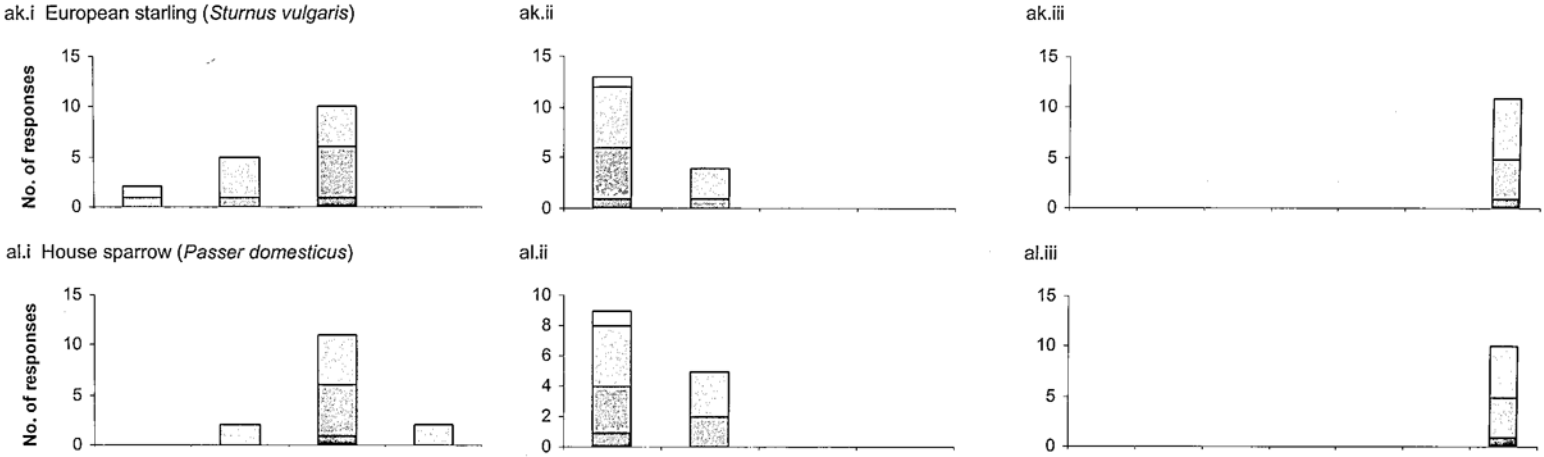


Figure A1.1. (ak-al) The number of responses received for each option of the survey questions. Black indicates the respondent was an expert, dark grey indicates they had high knowledge, light grey indicates they had average knowledge and white indicates they had low knowledge.

Table A1.2 Additional comments provided by the respondents.

Species name	Comment
Azure kingfisher (*Alcedo azurea*)	• Nests in holes in river banks. • Low records in Tasmania. • This species does not use hollows in Tasmania.
Sacred kingfisher (*Halcyon sancta*)	• Not a Tas. species, a rare vagrant.
Laughing kookaburra (*Dacelo novaeguineae*)	• Aggressive feral . • A pair nested on our property. We have never seen kookaburras attack the small birds and the latter are not afraid of the kookaburras. • Knew of 2 nests in hollows. Saw a competition with a brushtail possum where the kookaburra won. They are killed by vehicles but some people say a good thing.
Hardhead (*Aythya australis*)	• Wind farm risk during migration, loss of wetlands and hunting. • Numbers vary, if drought on mainland see more in Tasmania. • A rare vagrant. • Does it breed here in Tasmania?
Grey teal (*Anas gracilis*)	• Risk from windfarms, loss of wetlands and hunting. • An occasional visitor in low numbers.
Chestnut teal (*Anas castanea*)	• Risk from windfarms, loss of wetlands and hunting. • A lot breed in Tasmania, especially when rains maintain over spring into summer.
Pacific black duck (*Anas superciliosa*)	• Seems to be a survivor. Risk posed by hybridization with domestic ducks.
Australian wood duck (*Chenonetta jubata*)	• With Australian mainland drought cycles increasing this and other duck species may visit permanently thus displacing local populations. • V. rare in Tas in 1910, has increased rapidly during last 20 years. • Risk from windfarms, loss of wetlands and hunting. • Becoming a problem for farmers, eating grass etc and fouling up pastures. Nest in trees.
Australian shelduck (*Tadorna tadornoides*)	• Potentially at more risk from windfarms during migration also loss of wetlands and hunting. • A recent migrant to King Island. Good numbers breed here but do not stay after New Year. • Persecuted by ill informed persons - hunters and farmers.
Australian kestrel (*Falco cenchroides*)	• This species is rare on mainland Tas (>10 pairs) and is not known to use tree hollows (i.e. uses cliff sites). Larger populations on Bass Strait Islands but all nests known from cliffs and corvid nests. • Likely to be persecuted by ill informed persons.
Peregrine falcon (*Falco peregrinus*)	• Land clearing for development. • Human impacts greatest risk. Windfarms may pose a risk. • Predation by 'pigeon fanciers'. • Peregrines do not use hollows or other birds nests in Tas - cliffs only. • Often persecuted by ill informed persons.

Table A1.2 continued

Species name	Comment
Southern boobook (*Ninox novaesseelandiae leucopis*)	• Kookaburras, clearfelling. • Pesticide usage is of concern for population of this species and illegal trade in bird species. • Wood collection a problem. • Occasionally persecuted by ill informed persons.
Masked owl (*Tyto novaehollandiae castanops*)	• Also uses caves for breeding. • Pesticide usage is of concern for this species and illegal trade in bird species. • Breeds in hollows but roosts often in cliffs and vegetation. • Occasionally persecuted by ill informed persons.
Australian owlet nightjar (*Aegotheles cristatus*)	• Sometimes nest in fence posts. • Environmental concern re. use of pesticides etc. • Distribution of species confirmed in SW Tas through nest box usage.
Spotted quail thrush (*Cinclosoma punctatum*)	• Feral cats and maybe foxes are accounting for this bird. • Nest beside tree stumps. • Three years ago I observed a nesting female on 2 eggs. The nest was in a curled up piece of bark on the ground. • This species rarely uses true hollows. Occasionally in the base of tree cavities.
Grey shrike thrush (*Colluricincla harmonica*)	• One pair of grey shrike thrush nested in a manfern over summer in the backyard. The nest was a construction of shredded bark, twigs and fibre between fronds and fern and trunk. • Over the years have found more nests in the forks of scrubby trees and rarely in hollows. • Uses stumps for nests. • Urbanisation increasing threat. • Feral cat predation impacting on this species. • Mostly uses open cavities, shelves rather than hollows.
Welcome swallow (*Hirundo neoxena*)	• Nest yearly under house eaves. • They seem to have enjoyed our civilisation. Used to nest on rock faces. Now like buildings and under bridges. • Windfarms may pose a risk during migration. • Studied swallows in Campania area. Never found any nesting in holes. Man made structures benefit their nesting. Use caves and cliff shelves. Land clearing helps them on their hawk over paddock trees when insect hatching. But never seen them over forests. Occasional road kill, especially juveniles.
Tree martin (*Hirundo nigricans*)	• Windfarms may be coming a risk during migration.

Table A1.2 continued

Species name	Comment
Yellow-tailed black cockatoo (*Calyptorynchus funereus*)	• Dependent on mature and rotting rainforest trees (esp. *Nothofagus cunninghamii*) for the fungi and grubs found in there. • Often get a small flock of 5-9 feeding in *Banksia marginate.* • It is against all my expectations to have ticked 'increasing' in this area. Until recent years, a flock of 10-15 would be as many as we would see. For the last 3 years have seen up to 80 in a flock. They have learnt to feed on *Pinus radiata* cones. • Wood cutting removes trees with potential nesting hollows. • The increase in sulphur crested cockatoos, galahs and corellas is a concern for this species.
Gang gang cockatoo (*Callocephalon fimbriatum*)	• Rarely seen in Tasmania. • Land clearing for urban expansion and illegal trade in parrots and other bird species.
Sulphur-crested cockatoo (*Cacatua galerita*)	• Have observed being shot/poisoned due to the damage they cause to urban gardens. • Illegal poaching of young birds.
Galah (*Cacatua roseicapilla*)	• Aggressive competitor for nesting sites.
Little corella (*Cacatua sanguinea*)	• No comments.
Long-billed corella (*Cacatua tenuirostris*)	• No comments.
Swift parrot (*Lathamus discolour*)	• Window strike kill. • Possible competition with the introduced bumblebee? For eucalypt nectar? Noticed a decline in numbers on Maria Island where there are no forestry activities or land clearing. • Serious threat from human impacts and introduced ferals. • Does not nest here. Migrants passing through (King Island report) • Illegal poaching of young birds.
Orange-bellied parrot (*Neophema chrysogaster*)	• Wind turbines are of concern for the migration of this species from Tas. to Victoria and vice versa. • Recent breeding successes.
Blue-winged parrot (*Neophema chrysostoma*)	• Ground feeder, cats, habitat loss means a one-way spiral. • These were a regular sight during winter. Feeding on seeds in apple orchards in the West Tamar area. Have not been on our property for at least 20 years. • Possible increasing threat from wind farms during migration. • Vagrants, have been past breeding records (King Island report). • Illegal poaching of young birds. • Competition with starlings for nest sites, serious for years. • High numbers but declining?

Table A1.2 continued.

Species name	Comment
Green rosella (*Platycercus caledonicus*)	• Often killed by motor vehicles when feed on road sides. • Often killed as 'vermin' by orchardists and gardeners. Often killed as a result of window collision.
Eastern rosella (*Platycercus eximius*)	• Population has declined in N. Tas. more abundant in SE. • Urban expansion and related habitat destruction is of concern for this species. • Knew of 3 nests. All hollows in Campania area. 1 was taken over by starlings. Concerned that many are being caught in grape nets. Illegal poaching of young birds. • Competition with starlings for nest sites. • Occasionally killed as 'vermin' by orchardists and gardeners. Often killed as a result of window collision.
Musk lorikeet (*Glosopsitta concinna*)	• Bees and starlings may be of concern, flocks appear to be reducing in winter. • Have seen an increase in numbers in Launceston this autumn. • Environmental degradation by human activities and feral species. • Picked up 3 road kill this summer. Only know of nest in a hole in gum trees. In established suburbs where many mainland flowering gum trees have been planted. • They seem to be an increasing problem in fruit trees in outer Hobart areas. • Illegal poaching of young birds. • Occasionally killed as 'vermin' by orchardists and gardeners, often killed as a result of window collision.
Rainbow lorikeet (*Trichoglossus haematodus*)	• Vagrant to Tasmania. • Urban expansion and habitat destruction is of concern for this species. • This species is probably establishing from cage escaped birds and very localised.
Striated pardalote (*Pardalotus striatus*)	• Also nests in holes in banks and sometimes in buildings and pipes. Also gravel-soil heaps. • Over the 70 years of my interest in birds, this species has remained fairly stable. Nesting sites have changed from mostly in hollow trees to mostly in tunnels in drain banks. • Plenty around Buckland area, have seen nests built in behind stores and on bricks in walls. • Reduced number of hollows available in urban corridors and bush. Bees and galahs compete for nest sites and kookaburras starlings feral cats and sparrows.
Forty-spotted pardalote (*Pardalotus quadragintus*)	• Wildfire is a concern for Flinders island population. • The forty-spotted seems to be appearing in new habitats and doing OK.
Dusky robin (*Melanodryas vittata*)	• Was quite common 70 years ago. Sometimes nested in the fork of a fruit tree. Never see them here now. • Land use changes and environmental degradation may be pushing this bird towards a decline.

Table A1.2 continued.

Species name	Comment
Flame robin (*Petroica phoenicea*)	• Was the most plentiful robin 70 years ago in this area. For many years only see the rare family passing through. • Land use changes and environmental degradation in Tasmania and on the mainland seems to me to be responsible for a major decline in a formerly common bird. Windfarms sites may be a further risk during migration. • See quite a few spread over wide areas. The vast area netted for orchards plus their high density agriculture must reduce a lot of feeding ground. •
Dusky woodswallow (*Artamus cyanopteris*)	• These birds used to nest in fruit trees at times. • Risk from windfarms during migration. Land use changes and environmental degradation could easily push this bird to decline. • Observed three nests. Two in the fork of trees and one against a tree trunk partly protected by a large piece of peeling bark. •
European starling (*Sturnus vulgaris*)	• Have witnessed 'gangs' of this bird expelling a nesting pair of swift parrots from an *E. viminalis* at Deloraine and blue winged parrots at Ross (*E. pauciflora*). • Estimates in area around 300 (Ulverstone). • I have witnessed this species competing for nesting sites locally and harassing parrot species to leave tree hollows. • Very opportunist bird. Nests in hollows, top of fence posts, in buildings. Good year for corbi grubs, seen extra large flocks of juveniles in winter flocks. Feral so no concern. •
House sparrow (*Passer domesticus*)	• Urban dweller rarely seen in native forest. • Pest species. Causes concerns for other species. Congregates around the human that brought it. Mostly located in disturbed environment so less of an environmental impact. • Feral. Opportunist. Hollows, rose hip bushes, haystacks, buildings, holes in a bank. Difficult to say status. When the rosehips (*Bria bushus)* were removed, v. few birds remained. • Sparrow numbers have been increasing greatly until 2004 since then numbers in flocks seem reduced greatly.

Results and Discussion

Hollow use

Although some variability in the survey responses was found for certain species, generally the respondents agreed on the hollow-using status of the bird species considered. The results of the survey largely confirmed the results of scientific studies where literature was available (see Chapter 2 for more detail of the literature).

Most respondents agreed on the species that use hollows only very occasionally or perhaps not at all in Tasmania. These species are the house sparrow (Figure A1.1al), Australian kestrel (Figure A1.1j) and the peregrine falcon (Figure A1.1k). Although there were a range of responses for the azure kingfisher (Figure A1.1a), most respondents, including the expert, stated that they do not use hollows in Tasmania and are only found at very low numbers here. Similarly, although a range of responses were received for the grey shrike thrush (Figure A1.1p), the majority suggested they very rarely use hollows. A number of additional species were mentioned by the respondents as being known to use tree hollows very occasionally. These were the black currawong (*Strepera fuliginosa*), brown falcon (*Falco berigora*), scarlet robin (*Petroica multicolour*), spotted pardalote (*Pardalotus punctatus*), White's thrush (*Zoothera lunulate*) and the scrubtit (*Acanthornis magnus*). It was also clear that several of the species considered were vagrants and rarely occurred in Tasmania. These were: the gang gang cockatoo (Figure A1.1y) and the sacred kingfisher (Figure A1.1b). Although the rainbow lorikeet (Figure A1.1ae) was considered a vagrant by many respondents, reports indicate that it is now established in Tasmania (M. Holdsworth pers. comm.). The conclusion reached from this survey is that 29 of the species selected for this survey use tree hollows more than very occasionally.

Great discrepancy was exhibited on the hollow-using status of the dusky woodswallow (Figure A1.1aj), with the expert being the only respondent to state that this species is continuously dependent on hollows while the majority of respondents stated they were non-dependent or did not use hollows. The literature states that the dusky woodswallow can use hollows for breeding but that they also use stumps and roost behind bark (Sharland, 1958; Coulson and Coulsen, 1981). Reports were also received in this study of nests in the forks of trees (Table A1.2). Similarly, for the Australian wood duck (Figure A1.1hi), the more 'experienced' respondents stated they were dependent on hollows, largely for breeding, although other responses of non-use were also received. The only Tasmanian report found in the literature for this duck species indicated that they can nest in hollows or on the ground (Sharland, 1958).

For the southern boobook (Figure A1.1l), responses ranged from continuous dependent to non-dependent on hollows. A study done by Bell *et al.* (1997), indicated that they use hollows for nesting but are occasionally recorded using nesting boxes and other man-made structures. For roosting, southern boobooks do use tree hollows but will often use dense foliage, rocky clefts, caves or man-made structures (Bell *et al.*, 1997). For the Australian shelduck (Figure A1.1i), the majority of respondents indicated the species was either breeding dependent or non-dependent. The literature states that they use tree hollows for breeding but can also use holes in the ground (Sharland, 1958). Reports were also received of their using rock crevices and disused rabbit burrows on islands and in treeless areas (C. Spencer pers. comm.). The degree of dependency was also unclear for the masked owl (Figure A1.1m), with the majority of respondents indicating they were non-dependent, but others stating they were dependent to some degree. From the literature it appears that this species nests only in tree hollows but can roost in other locations such as cliffs, caves, vegetation and, occasionally, man-made structures (Bell *et al.*, 1997). However, one respondent stated that they can also use caves for breeding.

The majority of survey respondents suggested that the chestnut teal (Figure A1.1f.i) was non-dependent on hollows. The only Tasmanian report found in the literature for this species indicated that they usually nest in tree hollows, although they can also nest elsewhere (Sharland, 1958). This report by Sharland (1958) could be interpreted as non-dependency or as being dependent on tree hollows for nesting. For the welcome swallow (Figure A1.1q), the responses were either that they do not use hollows or they are non-dependent. In the literature it is stated that they do use hollows but no indication of frequency is given (Sharland, 1958). For the tree martin (Figure A1.1r), the majority of responses were that they were non-dependent, but some stated they use them for breeding and roosting or were breeding dependent. In the literature it was stated that they mostly use tree hollows for nesting but can use other sites (Sharland, 1958). The majority of respondents for the galah (Figure A1.1b) indicated they are dependent on hollows for breeding. In Western Australia the galah uses hollows mainly for breeding (Rowley, 1990). For the long-billed corella (Figure A1.1x), respondents indicated either a continuous dependency on hollows or that the birds were dependent for breeding. In the literature it was stated that breeding has not been confirmed in Tasmania although it is believed to occur (Brown and Holdsworth, 1992). For the forty-spotted pardalote (Figure A1.1ag) respondents largely indicated either dependency or non-dependency on hollows for breeding. In the literature it appears that the degree of dependency can vary, as Brown (1986) found most individuals used hollows while Woinarski and Bulman (1985) found alternative sites were more frequently used.

Population status

The information collected in this survey indicated that two of the species considered were rare vagrants. These were the gang gang cockatoo (*Callocephalon fimbriatum*) and the sacred kingfisher (*Halcyon sancta*).

The results of the current survey were generally supported by the literature for those few species where literature was available on the population status of the species. This is the case for the duck species (Game Management Services Unit, 2005), the sulphur-crested cockatoo and the little corella (Brown and Holdsworth, 1992; Coupland, 2000). A mixture of responses was received for the galah (Figure A1.1t), ranging from increasing to stable at low numbers. In the literature it was indicated they are likely to be at low numbers but gradually increasing (Brown and Holdsworth, 1992; Barrett *et al.*, 2003). For the blue-winged parrot, respondents indicated they were either stable at low numbers or declining (Figure A1.1aa). Reports in the literature were conflicting, with some studies indicating they were increasing (Brown, 1979; Brown and Wilson, 1982) while others suggested they have declined since European settlement (Green, 1983). For the eastern rosella (Figure A1.1ac), respondents indicated they were either stable at low numbers or declining. Green (1983) suggested they were declining. For the forty-spotted pardalote the majority of respondents indicated they are either stable at low numbers or declining (Figure A1.1ag). It was previously thought that this species was declining but recent work suggests they may always have been found at low numbers (Bryant, 1997). There was, however, one species for which the literature did not support the survey responses. Survey respondents suggested that the masked owl (Figure A1.1m) was either stable at low numbers or declining. Although this species is listed as endangered at the state level (Schedule 3 Tasmanian *Threatened Species Protection Act 1995*), there is no evidence of a decline in numbers (Bell *et al.*, 1997) although anecdotal reports have been received saying masked owls were more commonly seen in the 1940s and 1950s (Mooney, 1997).

Most of the respondents provided similar responses for a particular species. For example, there was some variation for the long-billed corella, but the majority of respondents indicated they were increasing. For the grey shrike thrush, the majority of respondents indicated they were stable, with only a couple suggesting they were declining. Similarly, the majority of the more experienced respondents indicated that populations of the southern boobook were stable, while a few respondents suggested they were declining. Given this response and the fact that southern boobooks are found in a number of reserves across the state (Bell *et al.*, 1997), it is suggested that populations of this species are stable.

However, there were some species for which the respondents gave very mixed results for population status. The differences in responses obtained in this survey may be due to several reasons. Firstly, it is expected that most respondents will have greater knowledge of their immediate vicinity and less on a broader geographical scale. It is possible that the degree of hollow use, population trends and threatening processes will vary between geographical areas in Tasmania. Secondly, although attempts were made to make the questions and categories of responses clear to the survey participants, there is still likely to be an effect of interpretation of the questions. For example, what constitutes a population at 'high' numbers and one at 'low' numbers may differ between respondents. One sighting of a bird breeding or roosting in an alternate location may be interpreted by some respondents as 'non-dependency' while others will still rate the species as being 'dependent' because the majority of sightings are from tree hollows. The species for which mixed responses were given are discussed below.

For the dusky woodswallow (Figure A1.1aj) and the tree martin (Figure A1.1r) some respondents indicated that the population status was stable at high numbers while others indicated that they were declining. For the sulphur-crested cockatoo (Figure A1.1u), responses ranged from increasing to declining, although the more experienced respondents tended to indicate either increasing or stable at high numbers. Great discrepancy in responses was received for the green rosella (Figure A1.1ab), from increasing to decreasing but with the majority of respondents indicating they are stable at high numbers. For the Australian shelduck (Figure A1.1i), again a great range of responses was received, but again, the majority indicating populations of this species are stable. A survey conducted by the Department of Primary Industries, Water and Environment confirmed that populations of Australian shelducks are stable (Game Management Services Unit, 2005) but no information on population status was found in the literature for the other species.

For the yellow-tailed black cockatoo (Figure A1.1s), the majority of respondents indicated they were declining, while a number of others, including the one expert, indicated they were stable. Only one respondent indicated they were increasing, stating that "It is against all my expectations to have ticked 'increasing'. In this area (Swan Point) until recent years, a flock of 10-15 would be as many as we would see. For the last three years we have seen up to 80 in a flock. They have learnt to feed on *Pinus radiata* cones as have their white tailed cousins in southwest Western Australia". Concern for this species has been expressed because their habitat is degraded by forest harvesting as they are dependent on large hollows for breeding (Wilson, 1981; Fox and Brereton, 2003).

Similarly, for the musk lorikeet (Figure A1.1ad), six out of fourteen responses to the survey indicated the species is declining, while only a single response (from the 'expert') said that the species is increasing. Reports in the literature as to their status are conflicting (Bryant, 2002; Barrett *et al.*, 2003). The majority of respondents also indicated that flame robins (Figure A1.1ai) are in decline, while a couple, including the expert, indicated they are stable at high numbers. Comparisons done between two major national bird surveys (Barrett *et al.*, 2002) suggests a nation-wide decline in flame robins and anecdotal reports also suggest a decrease in numbers around Hobart (see Newman, 2002). It was suggested that changes in rainfall patterns as a result of climate change may be cause for concern for this species (Newman, 2002). The suggestion of a decline in populations of the yellow-tailed black cockatoo, musk lorikeet and flame robins, although not from the experts, is of concern and warrants further investigation.

Threatening processes

In terms of threatening processes, the one considered to be of major concern for most species by the majority of respondents is forestry activities (although for six respondents this included the orange-bellied parrot which nests in southwest Tasmania where forestry practices do not occur, Figure A1.1z). Agriculture was also considered to be a major concern. The degree of threat perceived to be due to competition for nesting sites varied between species, being quite high for some such as the orange-bellied parrot and very low for others (although these were largely those species considered to be non-dependent on hollows).

The effect of predation was perceived as being relatively unimportant for most species, although was still considered to be important for species such as the rainbow lorikeet (Figure A1.1ae). The effect of cars was also variable in their perceived threat, being considered of relative importance to some species such as the Australian owlet nightjar (Figure A1.1n), but of little concern for the majority. The effect of hunting was greatest for the duck species, of which several species can be legally hunted (Game Management Services Unit, 2005). However, hunting was also considered to be of some importance for the owl species and the non-hollow-using peregrine falcon. One threatening process which was not provided as an option but was mentioned by several of the respondents was death to birds caused by windfarms.

The species which were universally ranked as having no threat were the introduced house sparrow (Figure A1.1al) and European starling (Figure A1.1ak). However, a number of other species were also considered to have no threats. Occasionally a respondent indicated there were no threats while simultaneously specifying threatening processes. This was interpreted

to mean that although the indicated threatening process does kill some individuals, it is not of major concern. Those species with the greatest responses of 'no threat' were the sulphur-crested cockatoo (Figure A1.1u), the pacific black duck (Figure A1.1g), the galah (Figure A1.1t), the Australian wood duck (Figure A1.1h), the grey shrike thrush (Figure A1.1p), the laughing kookaburra (Figure A1.1c), the welcome swallow (Figure A1.1q) and the rainbow lorikeet (Figure A1.1ae).

Conclusion

The responses to questions posed in this survey provided support to the fact that 29 bird species commonly found in Tasmania are likely to regularly use tree hollows for either roosting or nesting. The results from this survey indicate, however, that only one species, the Australian owlet nightjar, is considered to be dependent on tree hollows for both nesting and roosting. Nineteen other species are believed to be largely reliant on tree hollows for nesting, while the remaining 10 species use tree hollows to varying degrees. Four hollow-using bird species are currently listed as threatened in Tasmania (swift parrot, orange-bellied parrot, forty-spotted pardalote and the masked owl; Schedule 3 Tasmanian *Threatened Species Protection Act* 1995). Respondents to the survey expressed further concern over the status of the yellow-tailed black cockatoo and the musk lorikeet. The threatening processes considered to be of greatest concern are associated with land clearing (forestry activities and agriculture).

Appendix 2. Predictive growth model code for WinBUGS

```
Inits
list(sdtau=1,base=0, a=c(NA,0), b=c(NA,0.001),d=c(NA, 0.001,0.001,0.001,0.001), g=0.001, h=0.001)

model
{
for(i in 1:**) # replace "**" with the number of data points to be examined
{
X[i,1] <- (dbh[i]-avdbh) / sddbh
X[i,2] <- (soilp[i]-avsoilp) / sdsoilp
X[i,3] <- (avtemp[i]-avavtemp) / sdavtemp

predage[i] <- pred[i] * pred[i]
pred[i] ~ dnorm(mean[i], tau)
mean[i] <- base + g*X[i,1] + h*X[i,1] * X[i,1] + a[stand[i]] + m*X[i,2] + n*X[i,3] + b[asp[i]]
}

avdbh <- 101.39
avsoilp <- 255.55
avavtemp <- 10.41
sddbh <- 46.08
sdsoilp <- 134.54
sdavtemp <- 0.988

base ~ dnorm(15.05, sdbase)
sdbase <- 1/(0.2138*0.2138)
tau<-1/(sdtau*sdtau)
sdtau ~ dnorm(2.103,sdsdtau)
sdsdtau<- 1/(0.09026*0.09026)

a[1] <- 0
a[2] ~ dnorm(-1.091, sda2)
sda2 <- 1/(0.3381*0.3381)
a[3] ~ dnorm(-1.529, sda3)
sda3 <- 1/(0.3035*0.3035)

b[1] <- 0
b[2] ~ dnorm(-1.006, sdb2)
sdb2 <- 1/(0.4178*0.4178)

g ~ dnorm(3.644, sdg)
sdg <- 1/(0.1918*0.1918)
h ~ dnorm(-0.4625, sdh)
sdh <- 1/(0.0637*0.0637)
m ~ dnorm(-0.9028, sdm)
sdm <- 1/(0.1456*0.1456)
n ~ dnorm(-0.6233, sdn)
sdn <- 1/(0.1414*0.1414)
}
```

Data format:
Dbh [] soilp[] avtemp[] asp[] stand[]
Dbh, soil phosphorus and average temperature are continuous variables and aspect and stand are binary variables (aspect: 1 = N, E, S; 2 = W) (Stand: 1 = mature forest; 2 = mature with regrowth, 3 = regrowth with mature or regrowth).

Appendix 3. Details of the tree hollows and sites found to contain animal evidence that could be identified

Species	Number of records	Number seen in pre-fall survey	Entrance diameter (cm) (mean ± sd) [b]	Hollow depth (cm) (mean ±sd)[b]	Hollow aspect (°)	Tree age (years)	Tree diameter (cm)	Site altitude (m)	Stand age [a]	Comment
Trichosurus vulpecula	45	21	12 ± 7 (41)	43 ± 38 (37)	5N, 10E, 8S, 10W, 8UP	289 ± 137	138 ± 68	283 ± 158	17M, 8MR, 18RM, 2R	The hollows were located equally in the trunk and branches with branch diameter ranging from 13 to 200 cm. 42% of the hollows faced upwards compared to 30% of all hollows. The trees varied greatly in damage and structure.
Pseudocheirus peregrinus	7	2	14 ± 4 (4)	32 ± 25 (4)	N, 2E, 2S, W, UP	343 ± 129	187 ± 60	365 ± 128	2M, 4RM, 1R	One sample is believed to have been brought there by a bird so is not included. Hollows were located in a branch (except one) more than 16 cm in diameter. The trees were more than tree shape five, were more than 40 m tall (except one at 20 m), had at least 3 small and one large hollow. 3 of the used hollows were found in the one tree.
Petaurus breviceps	1	1	7	18	E	196	63	250	RM	A skeleton was found in a hollow in mid-north Tasmania. The hollow was located 20 m off the ground in a branch 32 cm in diameter. The tree was healthy (shape 3), 26 m tall and with minimal burn damage to the bark.
Certcartetus nanus	1	0	6	105?	E	238	176	273	M	Four individuals were found in one hollow. The hollow was in a branch 33 cm in diameter, located 36 m from the ground. The tree was approximately 74 metres tall, had a healthy crown (shape 3) and minor fire damage to the bark. Two medium-sized hollows had been seen pre-felling.

Appendix 3 continued

Species	Number of records	Number seen in pre-fall survey	Entrance diameter (cm) (mean ± sd) [b]	Hollow depth (cm) (mean ±sd)[b]	Hollow aspect (°)	Tree age (years)	Tree diameter (cm)	Site altitude (m)	Stand age [a]	Comment
Certcartetus lepidus [c]	9	9	NA	NA	NA	323 ± 158	180 ± 88	286 ± 84	5M, 1MR, 2RM, 1R	Two of the nests were found on one tree. The trees were mostly more than 40 m tall (with two exceptions at 26 and 29 m).
Bat spp.	10	4	14 ± 9 (9)	68 ± 38	2N, 5E, 1S, 1W	260 ± 72	156 ± 53	305 ± 106	2M, 2MR, 6RM	All bat evidence was hairs, with the exception of one scat. Three instances of use were confirmed to be a bat by the experienced hair identification expert and the rest not. Two were located in the trunk and the rest in the branches. Branch diameter ranged from 26 to 130 cm. Most trees were taller than 30 m (except one at 15 m) and had a range of hollows. Five showed severe burn damage.
Bird spp.	29	20	13 ± 8 (28)	67 ± 128 (24)	10N, 9E, 4S, 3W, 2UP	307 ± 135	157 ± 56	261 ± 130	12M, 3MR, 13RM, 1R	It was not possible to identify most birds to species level. 17 of the hollows were located on the trunk The trees varied greatly in shape and burn damage.

[a] M indicates mature forest, MR mature with regrowth, RM, regrowth with mature and R regrowth.

[b] Brackets indicate the number of hollows used if different from the total number of samples. ? denotes uncertainty in the precision of the estimate due to smashing of the hollow.

[c] The nesting sites of *C. lepidus* are not included in any other analysis as they were located below 2 m above the ground and were located in the bark of the tree rather than in a hollow.